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**Section 7.0**  
**VALUE ENGINEERING REPORT**

**REMEDIAL DESIGN FOR THE  
VIRGINIA WOOD PRESERVING SITE  
RICHMOND, VIRGINIA**

**VALUE ENGINEERING REPORT**

**NOVEMBER 10, 1994**

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## **1.0 INTRODUCTION**

The approved Remedial Design Work Plan (RDWP) work plan specifies that Value Engineering (VE) will be performed on various aspects of the remedial action for the purpose of attaining both greater efficiency and cost savings. When VE is performed that results in a design change affecting the environmental design criteria, documentation will be submitted summarizing the VE recommendation. This documentation will describe the possible alternatives, the potential savings and any implementation problems which might occur.

### **1.1 VE APPROACH**

The approach taken during this VE evaluation is to analyze those portions of the remedy at the site that appear to show potential for significant cost reduction or value added by the implementation of engineering and scientific rationale and controls. Value Engineering is defined as a process executed during design and construction that applies engineering and scientific rationale to specific aspects of design or construction for the purpose of enhancing the value of a given component or reducing its cost.

### **1.2 VE INTENT**

For the remediation of hazardous waste sites, this process is primarily intended to reduce the overall project cost. This is particularly important in situations where there is potential for a high cost remedial activity that yields little or no benefit to the environment or human health. This document provides background and identification of promising items in the RA that may merit VE. Based on discussions with the agencies, some of these items may be selected for detailed VE study.

## **2.0 SITE DESCRIPTION**

### **2.1 SITE DESCRIPTION AND HISTORY**

The Virginia Wood Preserving Site is located on 10 acres to the north-northwest of Richmond, Virginia, in Henrico County (Figure 1-1 of General Description). It is situated near I-95, 2.4 miles west of the Parham Road exit at the intersection of Oakview Avenue and Peyton Street (Figure 1-2 of General Description).

VPI owns 4.96 acres and, until October 1991, leased 5 acres from the Richmond Land Corporation, an affiliate of the Richmond, Fredericksburg, and Potomac Railroad Corporation (RF&P). Richmond Land Corporation owns the land adjacent to the site on the northern, eastern, and southern sides. CSX owns the railroad line on the western side of Oakview Avenue. Virginia Wood Preserving is currently in the process of acquiring much of this land from RF&P with the land transfer to occur in the spring of 1994.

Prior to 1974, the facility was owned and operated by Virginia Wood Preserving Corporation, which was owned by TaCo Corporation and Taylor-Colquitt Company. TaCo purchased Taylor-Colquitt's interest in 1965. Later, Taylor-Colquitt was succeeded by Southern Wood Piedmont Company, a subsidiary of ITT Grinnell. In 1974, Rentokil, Inc., purchased TaCo's stock. Both TaCo and Virginia Wood Preserving Corporation were subsequently merged into Rentokil. Rentokil later changed its name to Rentokil SupaTimber, Inc., and, in September 1989, changed its name to VPI.

Construction of the Virginia Wood Preserving plant began in 1956, and modifications to the plant have continued to the present. The first treatment cylinder was installed in 1956; wood treating operations began shortly thereafter and continued without interruption until January 1990, when all operations ceased.

The sole business at the facility has been treating wood with preservatives; however, the methods and chemicals used have changed over the years. Chemicals and compounds used on the site included mineral spirits, No. 2 fuel oil, chromium zinc



arsenate (CZA), copper chromated arsenate (CCA), fire retardant, creosote, pentachlorophenol (PCP), and xylene. The fire retardant is believed to have been a water-based solution of ammonium phosphate or ammonium sulfate. It may also have contained ammonium thiocyanate as a corrosion-inhibiting additive. Of the compounds in the fire retardant solution, only the ammonium thiocyanate was believed to have had any potential environmental significance.

In 1964, the plant added air drying of decking, creosoting of marine piling, and fire-retardant treating to its existing processes. Treatment with CZA was replaced by treatment with CCA in 1964; treatment with PCP ended in 1980; and creosote treatment ended in 1983. Treatment operations ceased altogether in January 1990.

Over the years, many of the facilities installed on the site were taken out of service or removed (Figure 1-3 of the General Description). Many of the facilities were in use until January 1990, including the aboveground storage tanks (ASTs), treatment room, treatment cylinders, concrete drip pad, concrete holding pond, shop, office, and assorted sheds. After VPI discontinued treatment operations in January 1990, a polyvinyl chloride (PVC) cover was installed over the drip pad to prevent stormwater from falling on the surface. Also, VPI constructed a roof over the concrete holding pond. In the spring of 1991, VPI arranged for the removal of all wood treatment equipment from the site. All eight ASTs and the three treatment cylinders were dismantled and disposed of off-site by a regulated hazardous waste contractor. A layer of clean compacted clay was placed over the area where the cylinders were located, to prevent infiltration and surface water transport of site-related constituents. Also, a roof was built over the former tank farm area.

Wastes from the early wood treatment operations were reportedly discharged to the blowdown sump north of the treatment cylinders. In 1963, at the request of the Virginia Department of Environmental Quality (DEQ) Water Division, the previous owners of the site replaced the blowdown sump with the present concrete holding pond and constructed a covered, unlined holding lagoon. The concrete holding pond was linked to the covered holding lagoon by an underground drain pipe (Figure 1-3 of the General Description). These two waste management features were operated under a Virginia NPDES discharge permit. The Virginia DEQ's request for the installation of

these features apparently occurred because of reported fish kills in Talley's Pond (Figure 1-4 of the General Description) on January 2, 1962, and on two previous occasions. The Virginia DEQ believed that a "chlorinated cresol" was involved, and reportedly traced this substance upstream of North Run Creek to the plant.

In 1974, with the construction of a new water treatment/preservative recovery system, the discharge of process wastewater ceased because the wastewater was continuously recycled. The underground drain pipe that connected the covered holding lagoon to the concrete holding pond was closed and apparently abandoned in place. However, details of testing, sampling, or the method of abandonment are unknown. Based on these changes, the Virginia DEQ issued a no-discharge certificate.

In 1976 or 1977, a batch of CCA precipitated in a process tank and was rendered unusable. The precipitation was reportedly caused by reduction of the chromium from the hexavalent state to the trivalent state. This batch of approximately 1,100 to 1,400 pounds of CCA was disposed of at the site by placing it in a pit (with alternating 6-inch layers of lime) located along the northern fence line in the northeastern quadrant of the Site.

In 1987, the entire contents (all water and visibly contaminated soil) of the covered holding lagoon were removed and transported to off-site treatment/disposal facilities. Clean closure was not attempted, and no soil or water samples were collected. Because the cover was not replaced, an open excavation containing a combination of rainwater and groundwater remains at the site of the former holding lagoon.

Throughout the operational history of the site, treated wood was stored in nearly all open areas on-site. The nearly 10-acre site is relatively level because it was backfilled and regraded to provide a working surface for the wood preserving operations. Major structures currently on-site are depicted on Figure 1-4 of the General Description and include an office, a garage (shop), the concrete drip pad (covered), the concrete holding pond (now covered), an unlined lagoon, and an open-sided storage shed. An infrequently used railroad spur, which terminates at the northeastern property line, bisects the site.

### **3.0 VALUE ENGINEERING ANALYSIS**

Three aspects of the selected remedy for the site have been selected for analysis as to their potential for value engineering. These aspects include:

- groundwater extraction;
- water treatment and disposal; and
- soils treatment

A brief discussion of the remedy aspect will be followed by Engineering Analysis and discussion.

#### **3.1 GROUNDWATER EXTRACTION**

##### **3.1.1 Remedy**

The site consists of two well defined aquifer units that are separated by a clay hardpan layer. The upper, or perched, aquifer extends from approximately four to seven feet below ground surface and is impacted directly by the defined site hot-spots (creosote, PCP, CCA contaminated areas). The lower or saprolitic aquifer extends from the hardpan to approximately 25 feet below ground surface and is impacted by contaminant transport through the hardpan.

The groundwater extraction system specified in the Record of Decision (ROD) includes the installation of vertical caissons at strategic points on-site with perforated laterals extending from the caissons on top of the bedrock layer and the hardpan layer. The intent is that these laterals will extract contaminated groundwater and Dense Non-Aqueous Phase Liquids (DNAPLs) to be delivered to an on-site treatment plant. It is believed that by extracting this liquid in combination with the installation of a RCRA cap and slurry wall around the perimeter of the site, that an inward groundwater gradient will result, preventing contaminant migration outside the slurry wall (i.e., outside the site).

### **3.1.2 Analysis**

An analysis of this proposed system indicates potential for economies to be attained through engineering modifications to the caisson system. Site conditions both at the present time and after completion of the cap and slurry wall indicate that there is considerable latitude in the ultimate design of the groundwater extraction system resulting in the same inward groundwater gradient.

Site groundwater monitoring indicates that both the perched and saprolitic aquifers are extremely low yielding, thus making the utility of extraction wells limited. Extraction wells would have the advantage of assuring that groundwater extraction is being performed at the surface of the hardpan and bedrock layers; however, the number of wells needed to provide appropriate coverage of these aquifers is excessive.

Based on results of a recent modeling of the site groundwater, based on the inclusion of the cap and slurry wall in the system (see section 3.3), it appears that groundwater in the perched aquifer will flow towards the north boundary of the slurry wall thus providing an excellent condition for the interceptor trench system for extraction of the perched groundwater. For the perched system, this may prove to be an economical and technically pragmatic alternative to installation of shallow laterals for groundwater extraction.

Regarding the deep (saprolitic) aquifer, it appears that the most practical extraction option remains the caissons and laterals. While extraction wells would most likely be more economical, wells would result in multiple breaching of the hardpan layer and would not provide the coverage of the aquifer that the laterals would afford.

## **3.2 WATER TREATMENT AND DISPOSAL**

### **3.2.1 Remedy**

The remedy in the Record of Decision for water treatment and disposal indicated the treatment of contaminated groundwater in an on-site treatment plant using activated carbon to treat organic constituents.

### **3.2.2 Analysis**

The Record of Decision is particularly non-specific regarding pre-treatment of groundwater or the effects that required stormwater collection and treatment will have on the water landing and treatment systems. Also not considered was that the production of contaminated sludge and activated carbon, which as F-listed wastes, will result in considerable regulatory, technical, and cost ramifications. The ROD also did not address the issue of dioxin both in the effluent and residuals. At the present time, the Commonwealth of Virginia has determined that the discharge requirements for an on-site treatment plant will require attainment of a treatment standard that is technically impossible to meet. Since the activated carbon will be contaminated with dioxin during this process, it cannot be regenerated and will require disposal at a high-cost, permitted dioxin incineration facility. In addition, since stormwater flows and treatment were not considered in the ROD (for this site, stormwater will be the majority flow), the cost of water treatment and disposal is far greater than that indicated in the ROD.

One option to be considered includes pretreatment of site water for metals, turbidity and suspended solids followed by disposal in the sanitary sewer. Although technically and economically pragmatic, this option has run up against stiff opposition by Henrico County because of a county prohibition on discharge of stormwater or combined flow to the sanitary sewer. Another option for water treatment includes the use of an Ultra violet/Oxidation process for organic treatment rather than the exclusive use of carbon adsorption. This option would result in the use of a destructive technology for organics removal while greatly reducing the generation of dioxin containing F-listed

carbon. Finally, serious regulatory consideration could be given to allowing on-site disposal of generated sludge and carbon, thus reducing the cost and hazards associated with off-site disposal.

### **3.3 SOIL TREATMENT**

#### **3.3.1 Remedy**

Three specific areas on-site have been identified for treatment of soils above the hardpan layer. These areas include soils within 25 feet of the process drip pad, blowdown sump and the existing site pond as well as a fill area on the southeast portion of the site. The total volume of soil to be treated is approximately 12,850 cubic yards based on design modeling. The selected remedial technology for soil treatment is Low Temperature Thermal Desorption coupled with a non-combustive, recovery type air pollution control system.

#### **3.3.2 Analysis**

This item is quite complex and considerable regulatory and engineering rationale is involved in the analysis. The analysis is broken down into the following components:

- soil contaminant conditions/fate and transport analysis;
- risk analysis; and
- air pollution control system analysis.

##### **3.3.2.1 Soil Contaminant Conditions/Fate and Transport Analysis**

This section discusses the fate and transport of organic contaminants at the Virginia Wood Preserving Site. A three dimensional groundwater flow and transport model was constructed for the purpose of determining the concentrations of organic contaminants outside the site boundary after implementing the mandated remediation measures.

Construction of the flow and transport model also synthesizes all the relevant information available about the site, resulting in a better understanding of the physical and chemical processes and interactions occurring at the site.

The mandated remediation measures include a slurry wall around the entire site boundary, a multi-layer cap over the surface of the site, and groundwater extraction inside the slurry wall region. In addition, the "hot spots" of contaminated soils within the site boundary are to be excavated and treated using low temperature thermal desorption. The treated soil will then be backfilled. In part, the modeling study was conducted to determine the value of extracting the "hot spot" soils. Two remediation scenarios were modeled:

- Remediation Scenario 1: mandated remediation measures, including treatment of "hot spot" soils.
- Remediation Scenario 2: mandated remediation measures leaving "hot spot" soils in place.

The concentration of selected contaminants outside the slurry wall area were calculated using the model for each of the remediation scenarios. These concentrations were used to determine if there is any increased risk associated with leaving the "hot spot" soils in place over excavating and treating these soils.

In order to calculate the required concentrations, a three-dimensional flow and transport model was developed and calibrated, and this model was used to predict future concentrations. The development of this model proceeds through a series of four major steps.

1. Review of the data available for model construction and calibration.
2. Construction and calibration of the groundwater flow model using hydraulic head data.
3. Implementation of remediation scenarios into a flow model and predication of future water levels.
4. Implementation of remediation scenarios into a transport model and prediction of future concentration levels.

Each step depends on the results of the previous step. The reviewed data was used to construct and calibrate the steady state groundwater flow model. The remediation scenarios were implemented into the flow model, and a transient analysis was performed to predict water levels over a 30-year remediation period. Finally, the transient groundwater flow velocities calculated by the flow model were linked to the remediation transport model, and the concentrations over the 30 year remediation period were predicted. The details of the implementation of each step are described in the following sections.

#### **3.3.2.1.1 Data Review**

The data review was used to determine the physical and hydrogeological data available at the site for purposes of constructing the model. The sources reviewed were:

- 1988 Preliminary Investigation of Hydrogeologic Conditions and Soil and Groundwater Contamination at the Virginia Wood Preserving Site, Richmond, VA, Bennett and Williams, Inc.;
- 1990 Remedial Investigation Report, Virginia Wood Preserving Site;
- 1992 Phase II Remedial Investigation Report, Virginia Wood Preserving Site; and
- 1994 Remedial Design Work Plan, Virginia Wood Preserving Site.

A detailed description of the geology and hydrogeology of the site is contained in the sources listed above, and is not repeated here.

The data required for the construction of the models includes:

#### **Physical and geological data**

- surface elevations;
- elevations of the top of the hardpan layer;
- elevations of the top of the saprolite layer; and
- elevations of the top of the bedrock layer.



#### Information for determination of model domain and boundaries

- hydraulic head information in the vicinity of the site.
- creek location

#### Flow parameters

- hydraulic conductivity distribution for each layer;
- storage coefficient distribution for each layer;
- porosity distribution for each layer; and
- recharge due to precipitation to top layer.

#### Contaminant transport parameters

- source area;
- source strength;
- timing of source release;
- dispersivity of the porous media;
- sorption parameters; and
- present soil and groundwater contaminant distribution in all layers.

#### Parameters for predictive modeling

- slurry wall parameters (hydraulic conductivity);
- infiltration rate through cap; and
- extraction system configuration and pumping rates.

The data values used for modeling are described in the appropriate sections.

#### **3.3.2.1.2 Groundwater Flow Model**

The first step towards predicting the contaminant concentrations is to construct and calibrate a steady state groundwater flow model. This step is essential in order to determine the model parameter values, such as hydraulic conductivities, recharge rates, etc. This step is also very useful in understanding how the groundwater flow system works, and what components of the system have the most influence on the flow rates and directions at the site. The description of the groundwater flow model is divided into three parts. First, the conceptual flow model is characterized. Next,

the computer implementation of this conceptual model is described. Finally, the process of model calibration and the results are presented and discussed.

#### **3.3.2.1.2.1 Conceptual Model for Current Groundwater Flow System**

Four main hydrostratigraphic units are present at the site. The base of the flow system is formed by a low permeability unweathered granite bedrock unit. Above the bedrock is a saprolite unit that is from 4 to 14 feet thick, and acts as an aquifer. A hardpan unit ranging from 0 to 10 feet in thickness overlies the saprolite, and acts as a confining layer. The uppermost unit is a perched aquifer, with saturated thicknesses of 0 to 4 feet.

North Run Creek, approximately 150 feet north of the site, forms the only natural boundary for the flow system. The other model boundaries are determined from the hydraulic head distribution in the vicinity of the site.

These four hydrostratigraphic units contain two aquifers: the perched aquifer and the saprolite aquifer. Over much of the site, these aquifers are separated by the hardpan unit. However, the hardpan unit has two "holes" which allow connection between the aquifers. One of these "holes" is natural, caused by a thinning of the hardpan unit to zero. The other "hole" is man-made, caused by the installation or removal of the blowdown sump. Flow in both aquifers is generally from southwest to northeast.

##### **Perched aquifer**

The hydraulic head distribution and the flow directions in the perched aquifer are primarily controlled by the elevations of the top of the hardpan unit. The hydraulic head is typically one to three feet above the top of the hardpan layer, and flow directions are from high head to low head. Secondary controls on the head distribution in the perched aquifer are the distribution of hydraulic conductivity of the hardpan unit, the distribution of hydraulic conductivity of the perched aquifer, and areal recharge distribution.

The hydraulic conductivity distribution of the hardpan unit follows the soil types. In areas where the Colfax soil series is present, the hardpan is highly cemented and has very low permeability. In other areas, the hardpan has a somewhat higher permeability, allowing more hydraulic communication between the aquifers. In the areas of "holes" in the hardpan, the permeability is equal to the aquifer permeability, and the aquifers are directly connected. This hydraulic communication variation within the hardpan results in deviations of the hydraulic head distribution in the perched aquifer from the expected head distribution due to elevation of the top of the hardpan.

The distribution of flow in the perched aquifer may also be controlled by the hydraulic conductivity distribution within the perched aquifer. However, no evidence of a systematic variation in the hydraulic conductivity within the perched aquifer was found, so a uniform hydraulic conductivity is assumed.

Recharge also varies over the site, and areas of higher recharge result in higher hydraulic head values. Recharge over the model area can be divided into three zones. The highest recharge occurs over the unimproved forested areas. Medium recharge occurs over the wood preserving site, due to the compacted nature of the near surface soil and the stormwater runoff system. The lowest recharge occurs on the permanent wetland areas and the areas of groundwater discharge.

#### Saprolite aquifer

The saprolite aquifer is a confined aquifer in the vicinity of the site, and therefore the hydraulic head distribution is controlled primarily by the regional gradient. Under the regional gradient, groundwater flow in the saprolite aquifer is generally from southwest to northeast. Local deviations from the regional gradient occur due to the hydraulic communication with the perched aquifer through the hardpan layer, and local variations in hydraulic conductivity within the saprolite unit.

In regions where the hardpan unit has low permeability, the hydraulic head in the saprolite aquifer is not influenced by the hydraulic head in the overlying perched aquifer. In regions where the hardpan unit has higher permeabilities, the hydraulic head in the saprolite aquifer is somewhat influenced by the hydraulic head in the

overlying perched aquifer. In regions where a hole exists in the hardpan unit, the hydraulic head in the saprolite aquifer should be similar to the hydraulic head in the overlying perched aquifer.

The hydraulic head map of the saprolite aquifer presented in Figure 3-9 of the 1992 Phase II RI shows a large deviation from the regional gradient in a band across the center of the plant site. This large deviation is unusual in a confined aquifer. Possible causes of this deviation are (1) a high permeability channel in the saprolite unit; (2) unconfined conditions with the deviation caused by variations in bedrock elevations; (3) a low permeability channel "damming" the water behind the low permeability region. Large variations in the hydraulic conductivity of the saprolite do not generally occur, since the depositional environment was relatively uniform. The bedrock elevations do not reflect a high in the region of the high in the hydraulic head of the saprolite aquifer. The cause of the large deviation from the regional gradient is not understood conceptually.

#### **3.3.2.1.2.2 Implementation of Conceptual Flow Model**

Groundwater flow at the Virginia Wood Preserving Site was modeled using MODFLOW (USGS, 1984). The modeled domain contains the Virginia Wood Preserving Site and the surrounding area. Figure 3-1 shows the limits of the modeled domain and the model grid. The northern boundary of the domain coincides with North Run Creek. The other boundaries are chosen to encompass as much data as is available about the site, and are located as far from the site as data is available.

Within the model domain, the hydrogeology is discretized into three layers. Layer 1 represents the perched aquifer, layer 2 represents the hardpan unit, and layer 3 represents the saprolite aquifer. The elevations for the bottom and top of each of the three layers were calculated from the 1992 Phase II RI Figures 3-4, 3-5, and 3-7. Within each grid cell, an average bottom and top elevation for each layer was interpolated from these figures.

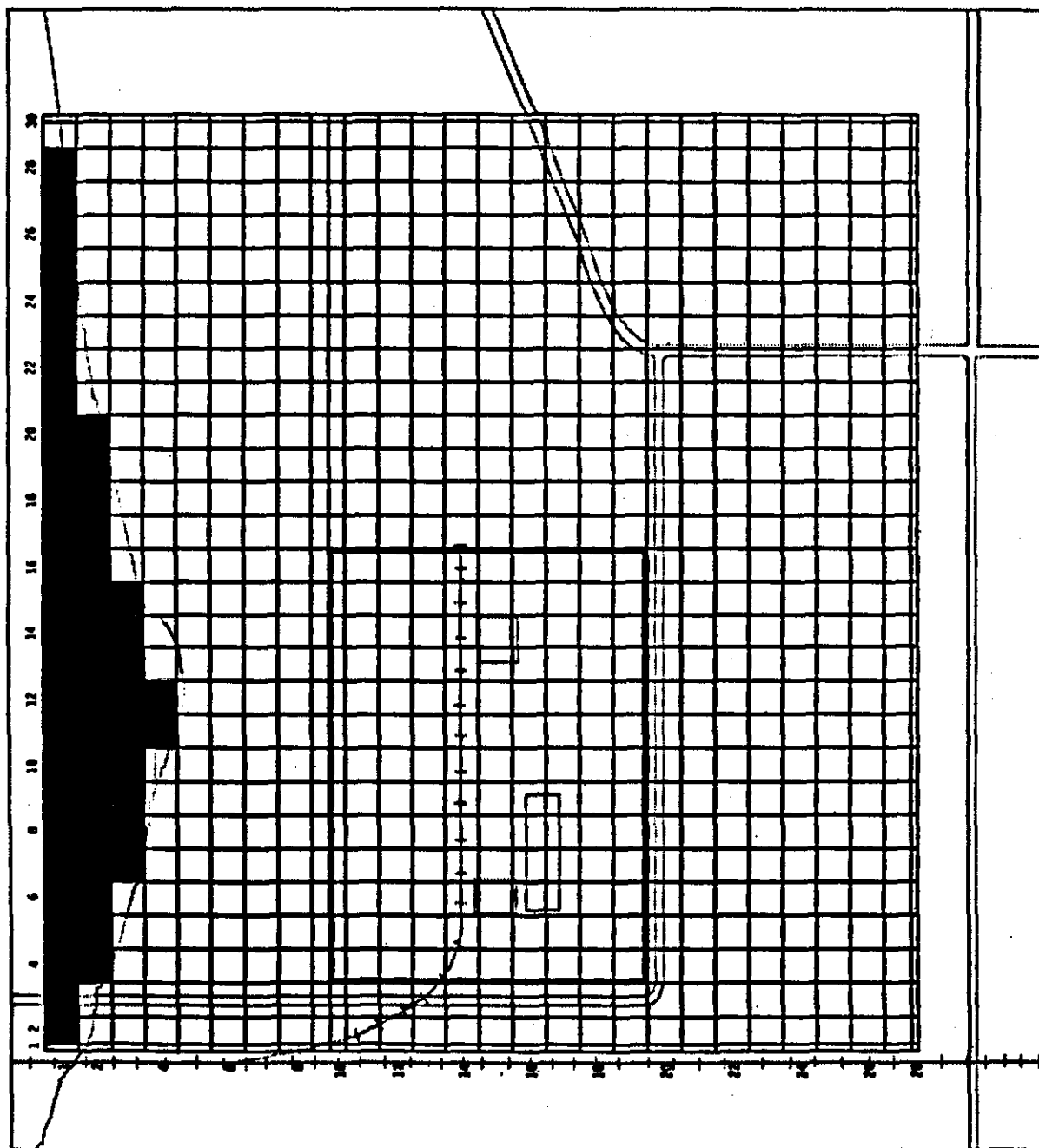


FIGURE 3.1  
MODEL GRID  
VIRGINIA WOOD PRESERVING SITE

A flow boundary condition is required for each of the model boundaries. The northern boundary of the model domain is imposed through a combination of no-flow boundaries and drain cells. North Run Creek is considered a groundwater divide, so a no-flow boundary is imposed along the creek. Along the top layer, North Run Creek was modeled using the drain package in MODFLOW. The average creek elevation within each cell was used as the drain elevation for that cell.

The western, southern and eastern model boundaries are considered constant head boundaries. These boundaries were placed as far as possible from the site so that the imposed head boundaries do not overly influence the calculated flow pattern within the site. The head values at the boundaries were calculated from the water level information obtained in the 1992 Phase II RI. Phase II RI Figures 2-10 and 2-11 were used to estimate the average head values within each cell along the constant head boundaries for the perched and the saprolite aquifer, respectively.

The parameters required for each grid cell are the hydraulic conductivity, and the top and bottom elevations. Recharge due to infiltration is also required over the top layer.

All model cells representing the perched aquifer were assigned a uniform hydraulic conductivity. Although the hydraulic conductivity undoubtedly varies within the perched aquifer, there is no evidence of a systematic pattern for the hydraulic conductivity distribution. The perched aquifer was assigned a uniform hydraulic conductivity of 0.173 ft/day, based on the average hydraulic conductivity obtained by Bennett and Williams using field and laboratory tests. During model calibration, this hydraulic conductivity value was increased to 0.75 ft/day. The increase in model hydraulic conductivity over measured hydraulic conductivity is consistent with the results of many modeling studies. A field test for hydraulic conductivity samples a much smaller volume than the model domain. This smaller volume may miss some large scale features that the model domain includes. The model scale hydraulic conductivity is therefore generally larger than the hydraulic conductivity measured using field tests.

The saprolite aquifer was also assigned a uniform hydraulic conductivity, for the same reasons described above. The assigned hydraulic conductivity for the saprolite aquifer

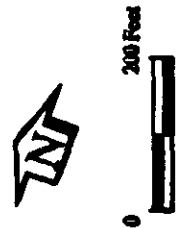
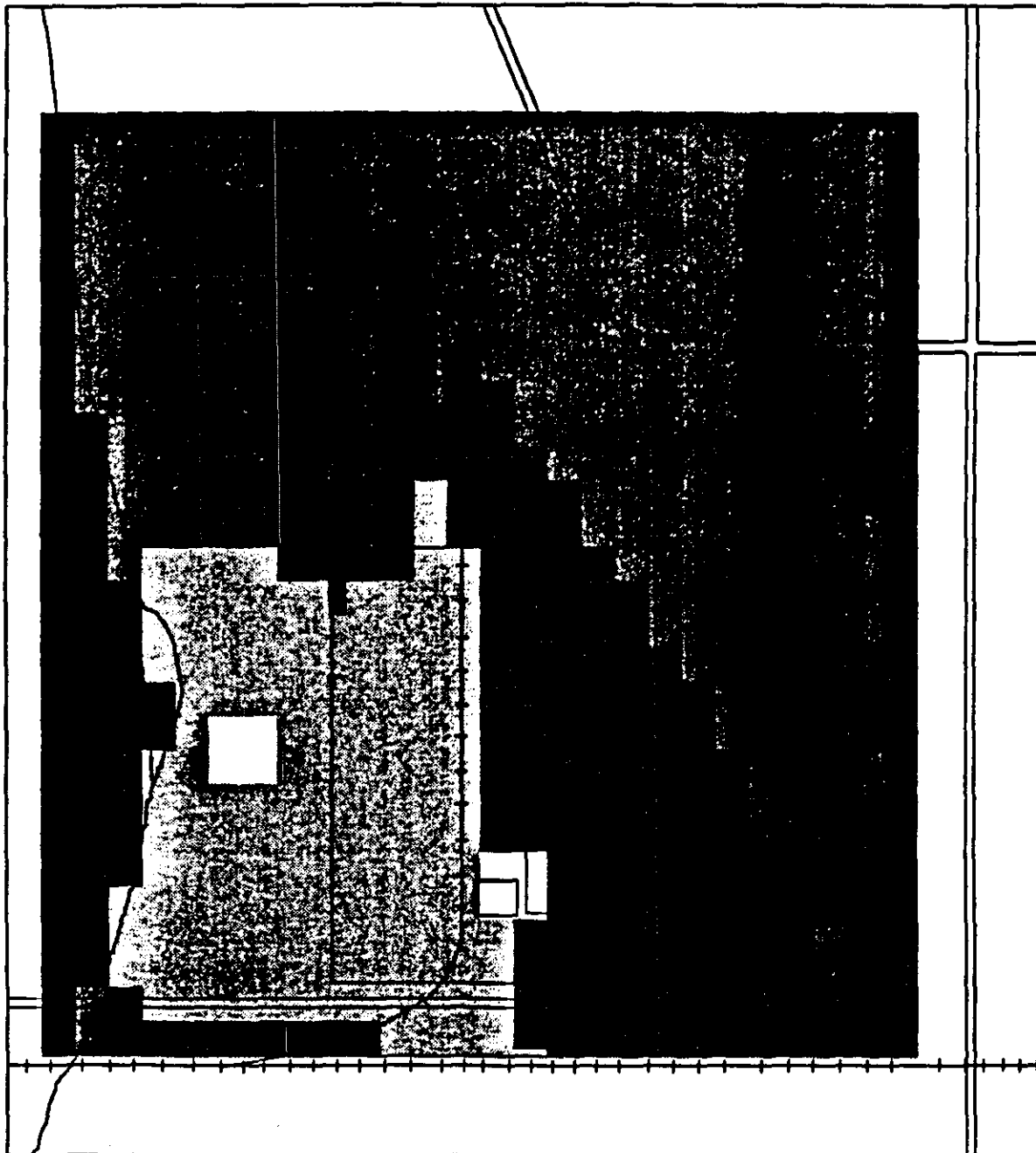
is 0.123 ft/day, based on the average hydraulic conductivity obtained by Bennett and Williams. During model calibration, this hydraulic conductivity value was increased to 0.75 ft/day.

The hydraulic conductivity within the hardpan unit shows a systematic variability, based on the soil type and the presence of "holes" in the hardpan. Figure 3-2 shows the modeled spatial distribution of the three zones representing the hydraulic conductivity within the hardpan. Zones H1 and H3 represent the hardpan in the vicinity of the Colfax soil series, with a laboratory measured hydraulic conductivity of 0.000028 ft/day. Zone H2 represents the hardpan under the other soil units, which was not measured in the laboratory. "Holes," or soft spots in the hardpan, also exist, with a laboratory measured hydraulic conductivity for the soft spots of 0.0028 ft/day. Using MODFLOW, a layer cannot be assigned a zero thickness, so the "holes" in the hardpan are modeled using a finite thickness layer with a hydraulic conductivity equal to the hydraulic conductivity of the adjacent layer. During model calibration, the hydraulic conductivity of H1, H2, and H3 were adjusted to 0.00005, 0.028, and 0.0005 ft/day respectively. Since the measured hydraulic conductivities are based on core analysis in the lab, an order of magnitude increase between the laboratory and model hydraulic conductivities is not uncommon.






Figure 3-3 shows the spatial distribution of the three recharge zones within the model domain. The unimproved forested areas receive the highest recharge, with a calibrated value of 2 inches per year. The wood preserving site receives a moderate recharge, with a calibrated value of 1 inch per year. The groundwater discharge areas receive no recharge. These calibrated recharge values are somewhat lower than expected, but the recharge rates cannot be increased significantly without using unrealistically high hydraulic conductivity values.

#### **3.3.2.1.2.3 Flow Model Calibration**

The flow model, as constructed above, must be calibrated in order to determine the model parameter values and be useful as a predictive tool. During the calibration process, measured hydraulic head data at various points throughout the model domain

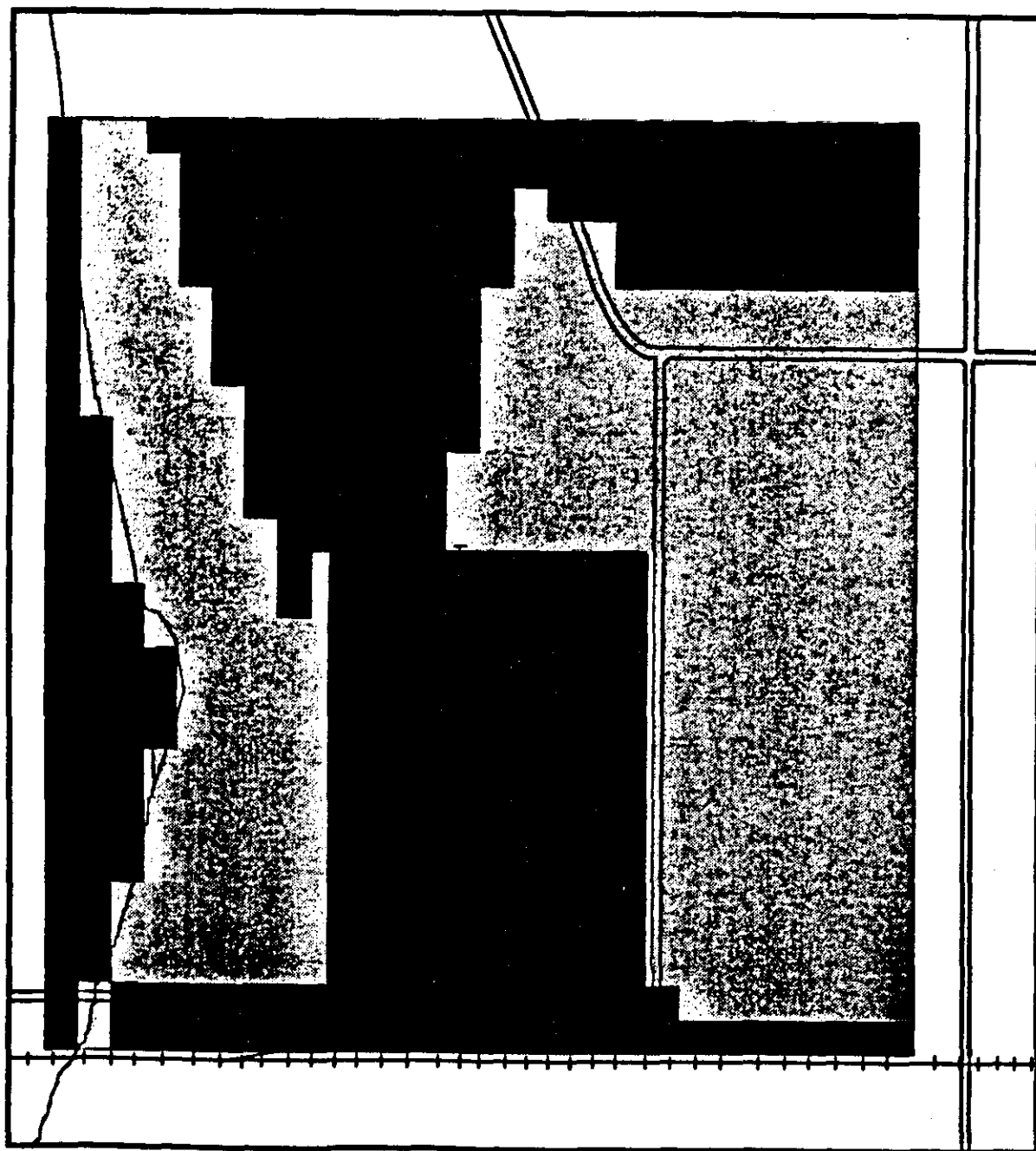


**Legend**

-  No Flow Cells
-  Hydraulic Conductivity Zone H1
-  Hydraulic Conductivity Zone H2
-  Hydraulic Conductivity Zone H3
-  Holes in Hardpan

**FIGURE 3.2**  
**HARDPAN HYDRAULIC CONDUCTIVITY ZONES**  
**VIRGINIA WOOD PRESERVING SITE**





**FIGURE 3.3**  
**RECHARGE ZONES**  
**VIRGINIA WOOD PRESERVING SITE**

are compared to the modeled hydraulic head data at those points. The model parameter values, such as hydraulic conductivity and recharge, are adjusted until the measured values closely match the modeled values.

The flow model was calibrated using the July 1992 water level measurements. Data are available for 14 wells in the perched aquifer, and 23 wells in the saprolite aquifer. Table 3-1 presents the well information, including the well name, the aquifer it is screened in, the July 1992 measured water level, the modeled head for the final calibrated model, and the residual (measured head minus modeled head) for each well.

During the model calibration process, all hydraulic conductivity parameters are systematically varied, both alone and in various combinations. The recharge parameters are also varied, both alone and in combination with the hydraulic conductivity parameters. After each run with a new set of parameter values, three checks are conducted. First, calibration statistics such as residual mean, residual sum of squares, residual standard deviation, etc., are calculated for both aquifers individually and jointly. These calibration statistics are used to determine which set of parameters resulted in the best fit of the observed and modeled head data. Second, hydraulic head maps for both aquifers are constructed and compared to the observed hydraulic head distributions, to make sure that the overall flow pattern is being reproduced. Third, the residuals are plotted on the site map to determine whether the residuals are spatially correlated. The set of parameters which performs the best on all three checks is chosen as the calibrated model parameter values. Over 200 model runs were required to calibrate the flow model.

Figure 3-4 shows the modeled water table elevations for the perched aquifer, and Figure 3-5 shows the modeled potentiometric surface for the saprolite aquifer. These figures also include the plotted model residuals for each aquifer. These figures can be compared to the 1992 Phase II RI Figures 2-10 and 2-11. In the perched aquifer, the modeled water table contour pattern follows the pattern shown in RI Figure 2-10. A groundwater divide is clearly present running through the middle of the site. The dry areas simulated by the model closely follow the dry areas observed in the field. In the saprolite aquifer, the general groundwater gradient as well as a slight groundwater divide is correctly simulated. However, the large deviation from the general

TABLE 3-1

## CALIBRATION TARGETS FOR GROUNDWATER MODELING

Page 1 of 2

Well No.	Aquifer	Measured Head	Modeled Head	Residual
DM-15A	Perched	209.44	209.13	0.31
DM-19A	Perched	207.55	206.86	0.69
BW-3A	Perched	207.39	208.02	-0.63
DM-20A	Perched	208.36	208.71	-0.35
BW-11A	Perched	199.42	dry	
BW-2A	Perched	200.14	dry	
BW-10A	Perched	203.23	206.04	-2.81
DM-1A	Perched	205.63	204.98	0.65
SW-4	Perched	196.56	197.30	-0.74
DM-22A	Perched	205.10	204.84	0.26
DM-21A	Perched	202.96	203.43	-0.47
BW-8	Perched	206.25	203.90	2.35
BW-9A	Perched	203.73	201.54	2.19
DM-4A	Perched	205.80	207.19	-1.39
BW-14	Saprolite	208.27	207.96	0.31
DM-5	Saprolite	208.8	208.38	0.42
DM-4R	Saprolite	205.27	206.17	-0.90
DM-15	Saprolite	207.33	206.83	0.50
BW-11	Saprolite	199.31	200.99	-1.68
DM-19	Saprolite	205.80	205.46	0.34
DM-3R	Saprolite	206.34	205.62	0.72
DM-20	Saprolite	206.49	206.19	0.30
DG-7	Saprolite	208.17	206.11	2.06
DM-23	Saprolite	197.87	201.06	-3.19
DM-2R	Saprolite	196.85	201.73	-4.88
DM-16	Saprolite	205.90	203.96	1.94
DM-1R	Saprolite	203.21	204.72	-1.51
ET-1	Saprolite	195.49	197.7	-2.21
DM-22	Saprolite	198.76	202.27	-3.51
DM-21	Saprolite	201.35	203.79	-2.44

**TABLE 3-1**  
**(Continued)**

Page 2 of 2

Well No.	Aquifer	Measured Head	Modeled Head	Residual
BW-12	Saprolite	196.58	198.38	-1.80
BW-9	Saprolite	202.22	201.93	0.29
BW-13	Saprolite	204.21	204.12	0.09
ET-2	Saprolite	196.30	196.28	0.02
ET-6	Saprolite	198.27	197.68	0.59
ET-7	Saprolite	200.81	202.30	-1.49
BW-10	Saprolite	205.61	203.50	2.11

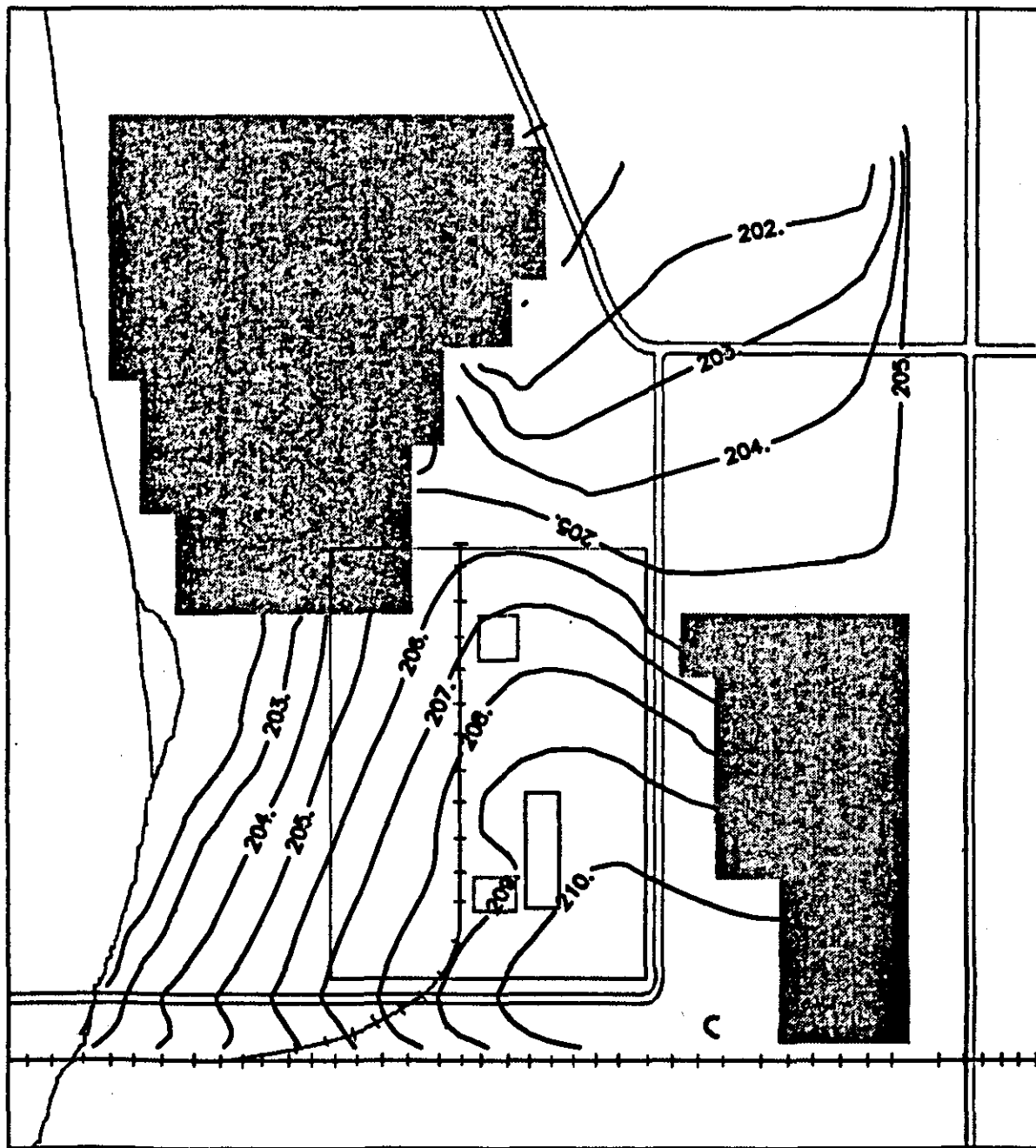


FIGURE 3.4  
PERCHED AQUIFER MODELED WATER TABLE ELEVATIONS  
VIRGINIA WOOD PRESERVING SITE

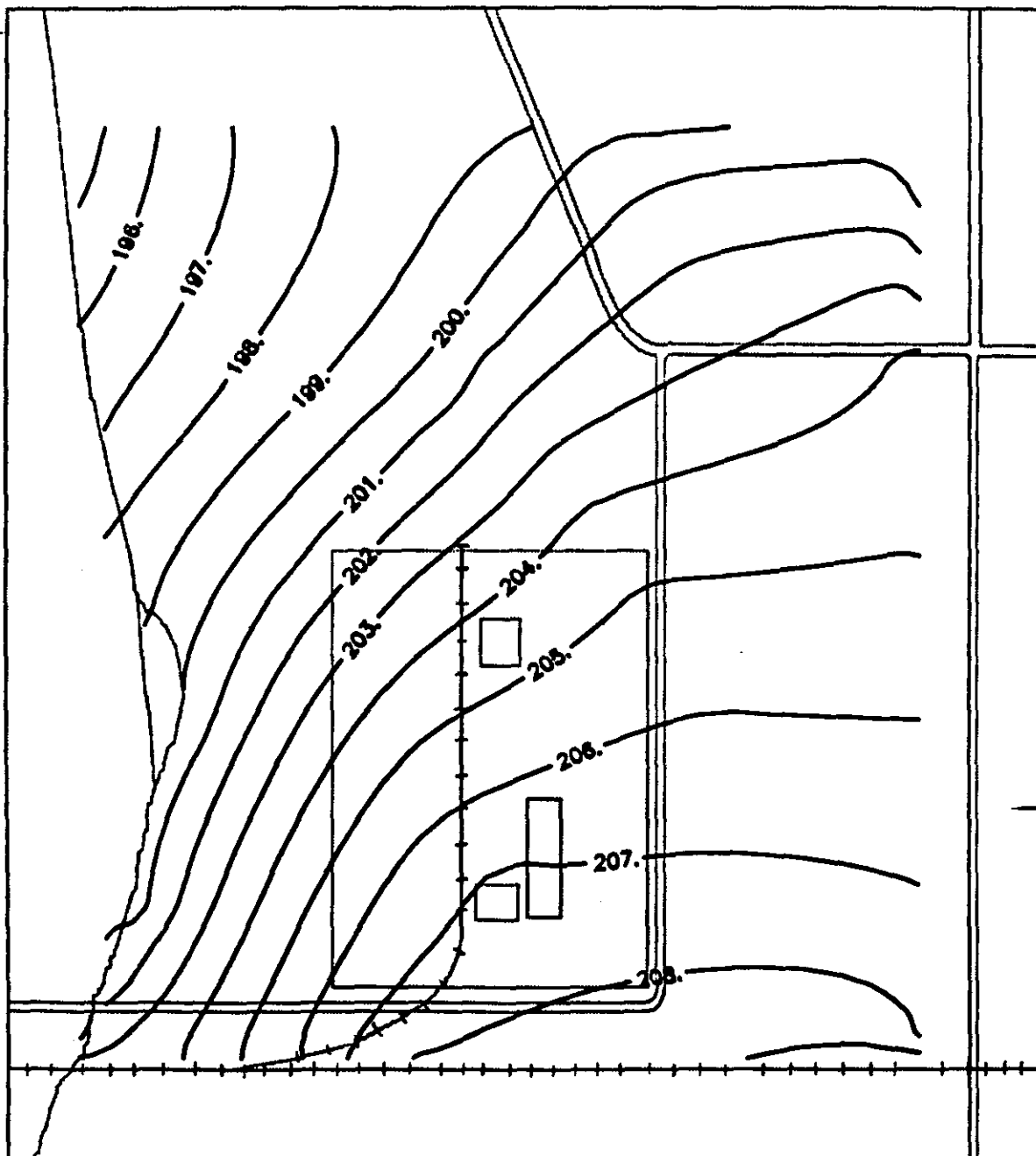


FIGURE 3.5  
SAPROLITE AQUIFER MODELED POTENTIOMETRIC SURFACE  
VIRGINIA WOOD PRESERVING SITE

groundwater gradient shown in the 1992 RI Figure 2-11 could not be simulated without using several zones with large hydraulic conductivity contrasts within the saprolite. These large hydraulic conductivity contrasts were not considered realistic, so they were not retained in the calibrated model. The calibrated model retains the flow characteristics within the saprolite aquifer, but not the magnitude of the deviation causing the groundwater divide.

#### **3.3.2.1.3 Conceptual Flow Model for Remediation**

After the steady state groundwater flow model was constructed and calibrated, the mandated remediation measures were built into the flow model. The changes to the flow model for remediation include placing a slurry wall around the site, reducing the recharge over the site due to a multi-layer cap, and designing an extraction system arrangement and pumping schedule for the interior of the site. A transient groundwater flow model was then run in order to predict the change in groundwater elevations over time after implementing the remediation measures.

A low conductivity slurry wall was simulated around the boundary of the site. The slurry wall extends from the ground surface to bedrock. The hydraulic conductivity of the slurry wall is specified to be 0.00028 ft/day, approximately one order of magnitude higher than the highest hydraulic conductivity obtained during the slurry wall trials in the site treatability study.

During remediation, the recharge distribution over the model area changes. The site is covered by a multi-layer cap, with a conservative estimated infiltration rate of 0.084 in/yr. Since the head on the interior of the site will be lower than the head on the exterior of the site, the former discharge areas (wetlands) may become recharge areas. The recharge in these areas are set equal to the wooded area recharge from the calibrated model during the period of remediation.

The extraction system on the interior of the site is included in the remediation measures in order to maintain inward groundwater gradients toward the slurry walled region. The actual remediation proposal is to include a set of caissons with horizontal

collectors as the extraction system. This type of extraction system is not easily simulated using MODFLOW. Instead, for modeling purposes, a set of wells inside the site was used to extract groundwater.

An extraction system is proposed for both the perched aquifer and the saprolite aquifer. However, the perched aquifer has a relatively low hydraulic conductivity and a saturated thickness of 0 to 3 feet. This combination of conditions creates difficulties for an extraction system in the system perched aquifer. At any significant pumping rate, the drawdown in the system quickly reaches the top of the hardpan, and the system cannot produce any more water. The drawdown cone from using wells would not extend very far from the well, so the wells do not have much influence on the general water table elevation. In addition, on the north side of the site, the top of the hardpan elevations are above the level of North Run Creek. This condition makes it difficult to create inward gradients along the entire northern boundary of the site. For these reasons, no extraction system was simulated in the perched aquifer. The extraction system in the saprolite aquifer, in combination with the hydraulic connection across the hardpan, was used to create drawdown in the perched aquifer. It is recognized that inward gradients will not be maintained around the entire site in the perched aquifer, but this condition is unavoidable.

Extraction wells are possible in the saprolite aquifer. Two extraction periods are required: an initial high extraction rate period to dewater the confined aquifer within the slurry walled area, and then a lower extraction rate period to maintain the drawdown conditions. Many arrangements of extraction well numbers, locations, and rates were tested. A suitable arrangement for modeling purposes includes nine extraction wells, spaced evenly within the slurry walled region. During the first year, these wells are pumped at an average rate of 60 ft<sup>3</sup>/day. During the remaining 30 year remediation period, the wells are pumped at an average rate of 8 ft<sup>3</sup>/day.

After modeling the above remediation measures, the groundwater flow model was run in a transient mode to predict the evolution of the groundwater levels over the 30 year remediation period. The initial groundwater levels were taken from the calibrated steady state groundwater flow model. Figures 3-6a, b, and c show the water levels in the perched aquifer at 1 year, 10 years, and 30 years after the start of remediation.



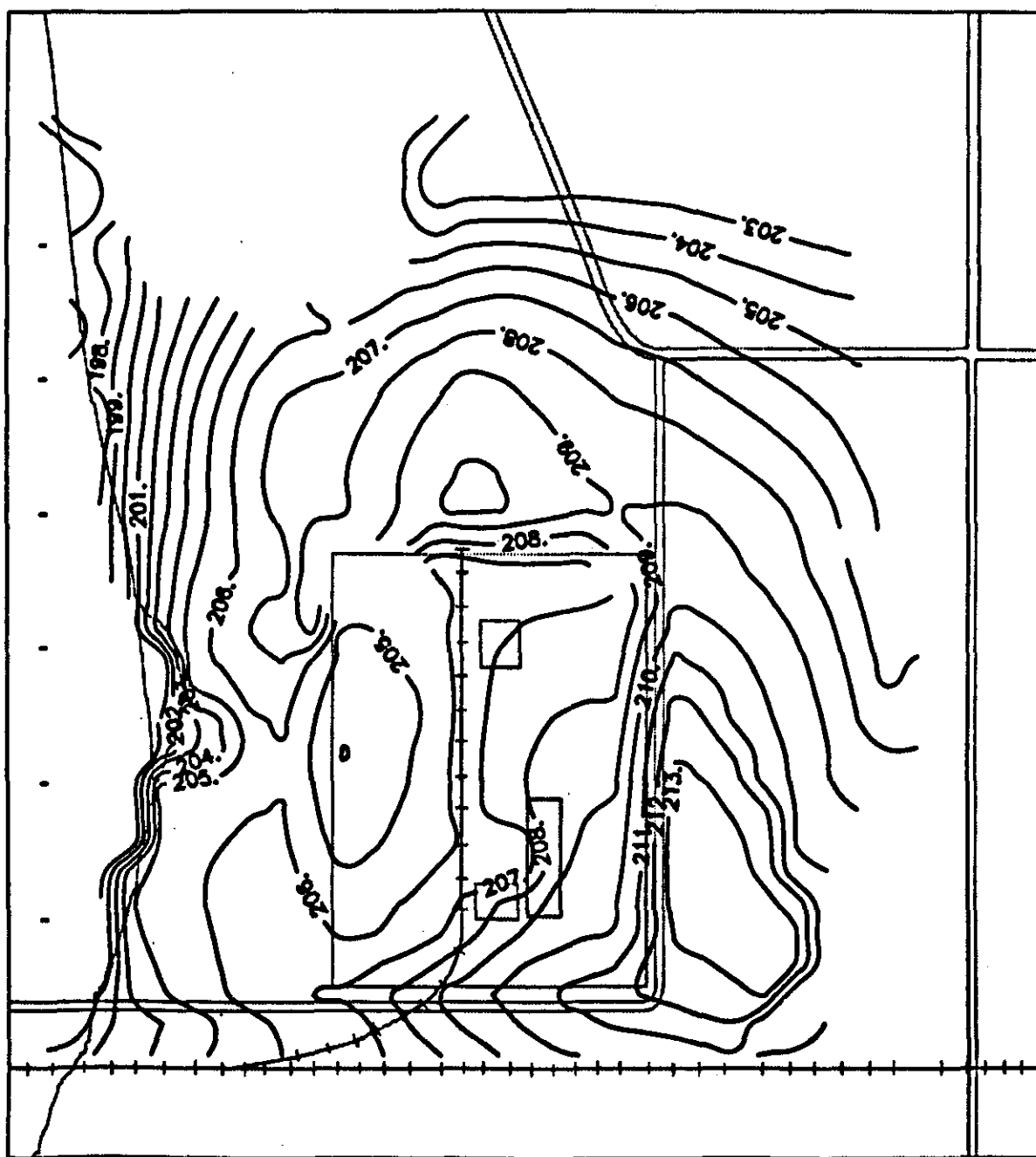


FIGURE 3.6a  
 MODELED WATER TABLE ELEVATIONS IN PERCHED AQUIFER  
 AT ONE YEAR AFTER THE START OF REMEDIATION  
 VIRGINIA WOOD PRESERVING SITE

AR304027

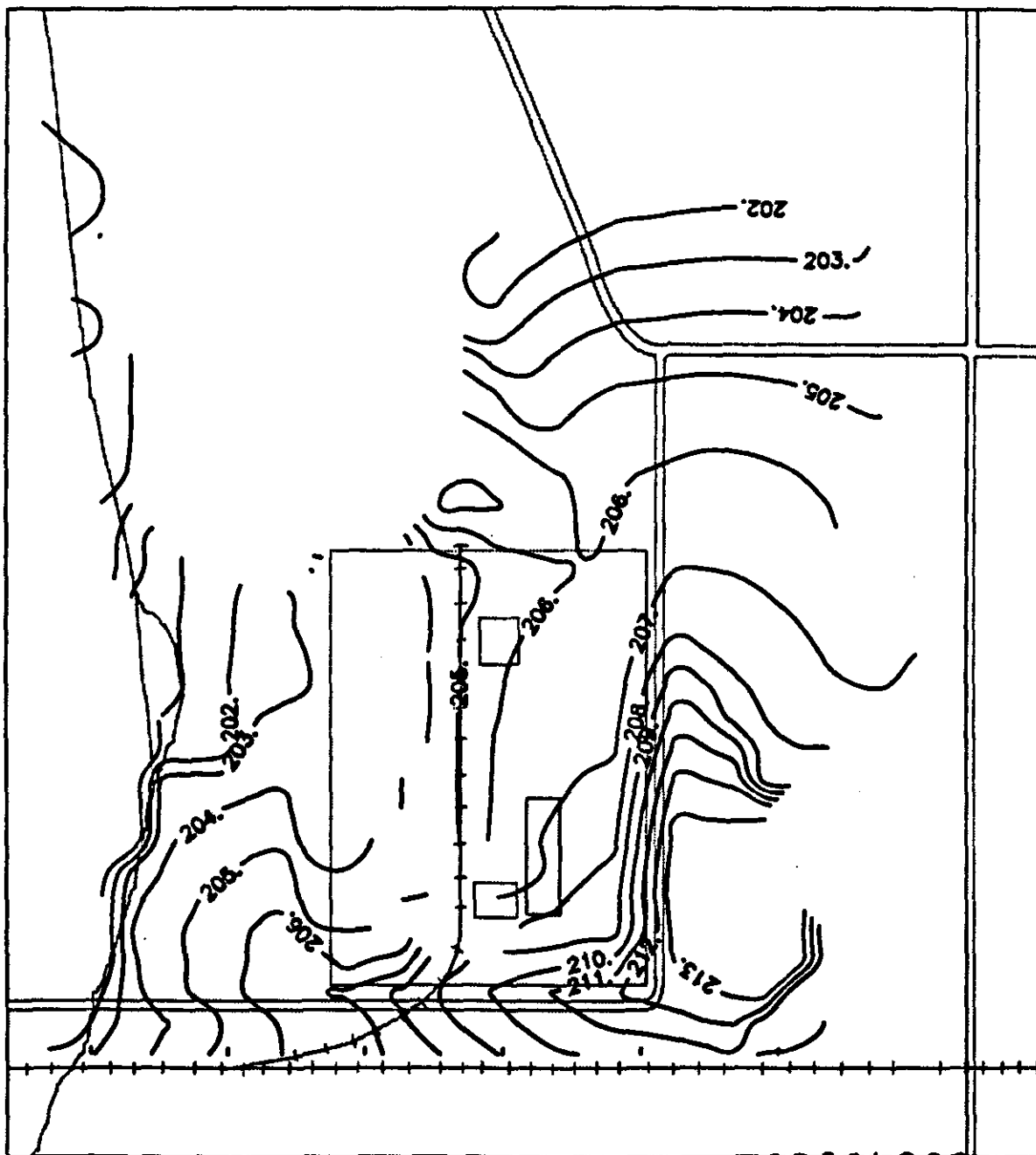
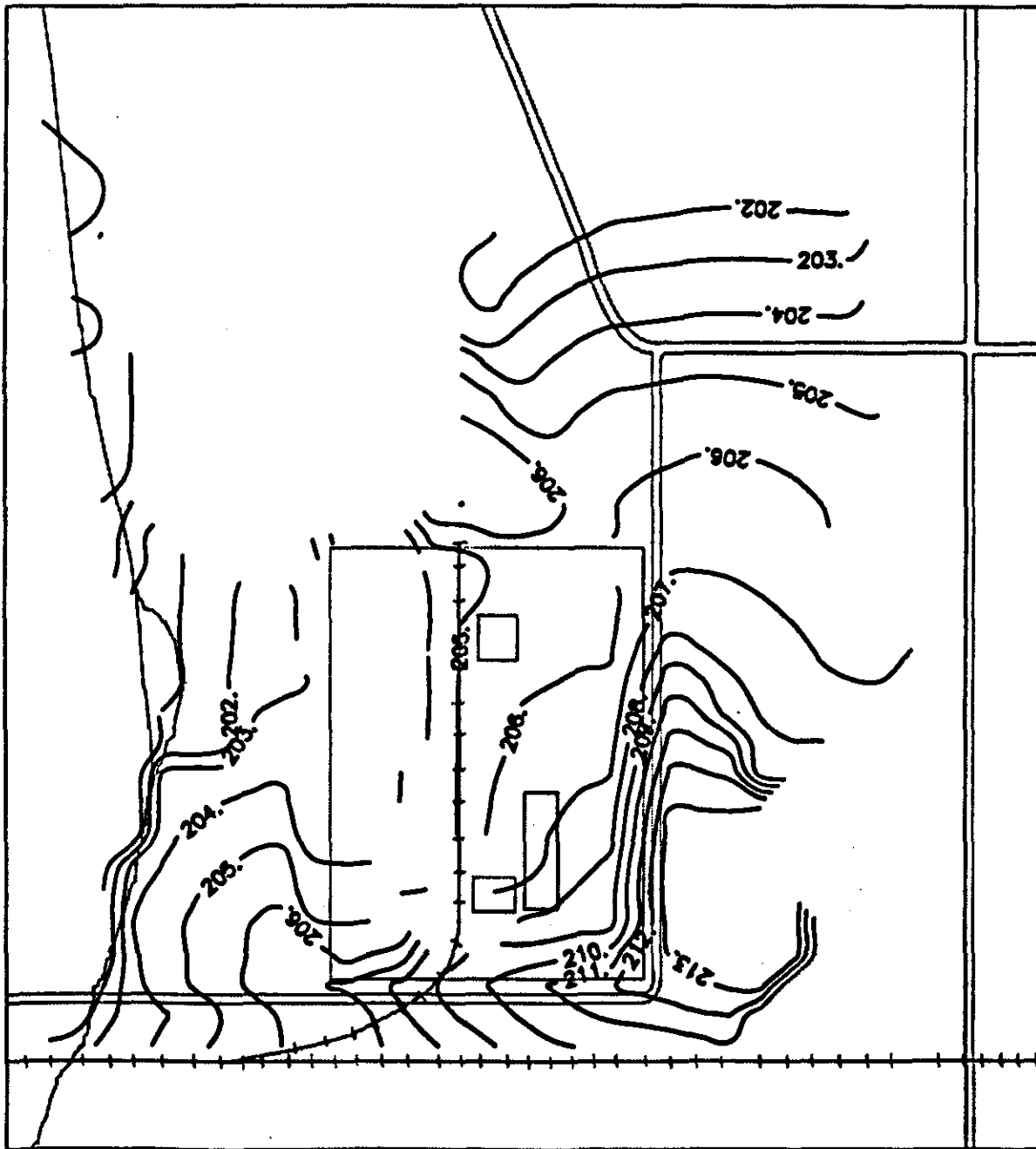


FIGURE 3.6b  
 MODELED WATER TABLE ELEVATIONS IN PERCHED AQUIFER  
 AT TEN YEARS AFTER THE START OF REMEDIATION  
 VIRGINIA WOOD PRESERVING

AR304028



AR304029



0 200 Feet

Legend

—204—  
Potentiometric Surface  
Contour (ft)

**FIGURE 3.6c**  
**MODELED WATER TABLE ELEVATIONS IN PERCHED AQUIFER**  
**AT THIRTY YEARS AFTER THE START OF REMEDIATION**  
**VIRGINIA WOOD PRESERVING SITE**

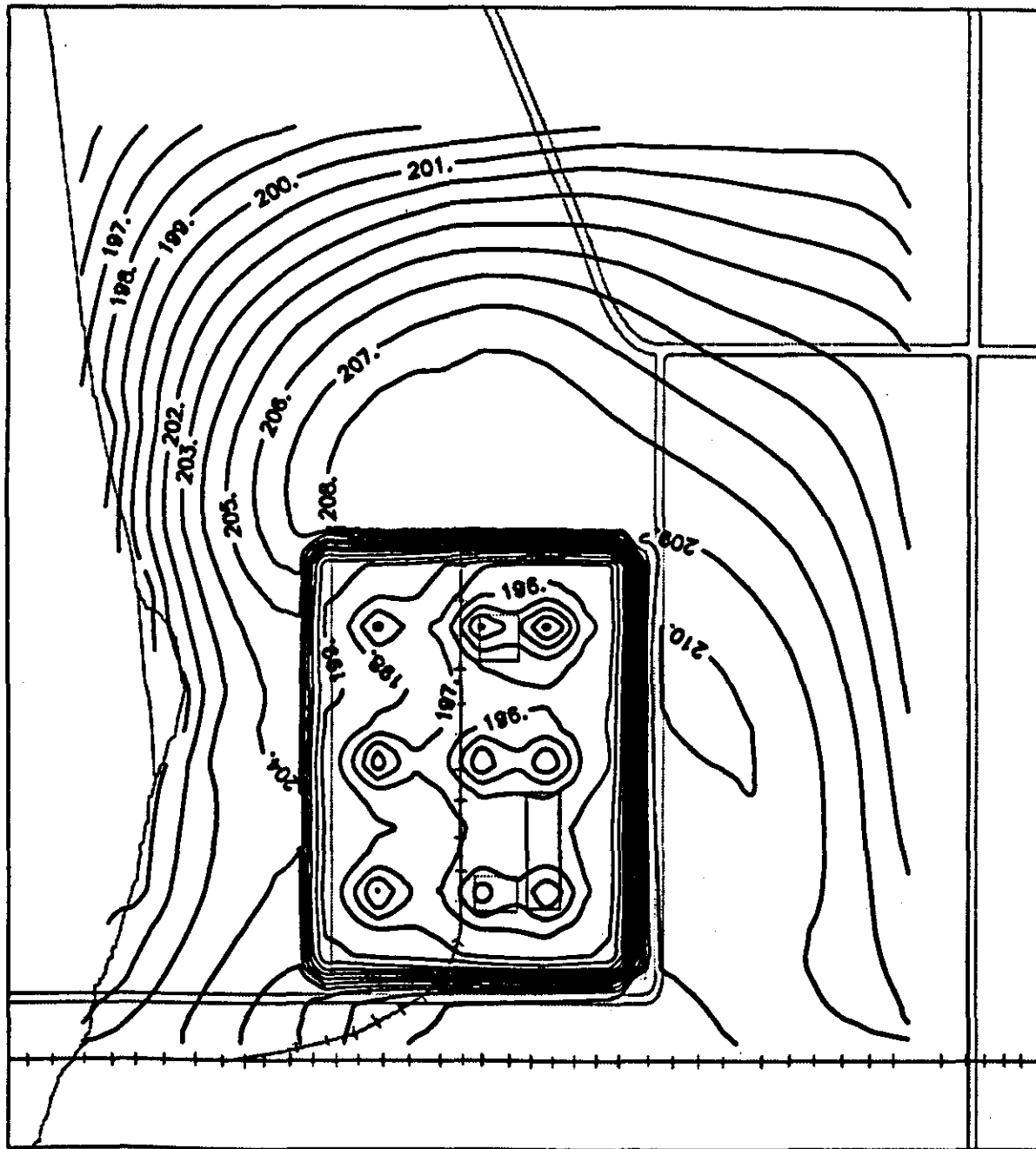
Several observations about flow within the perched aquifer can be made from these groundwater elevation maps.

- (1) Inward gradients are present around the south, west, and east side of the site during the entire remediation period.
- (2) Along the northern boundary, inward gradients are not maintained. Flow gradients are from the site toward North Run Creek.
- (3) Using this remediation scheme, the northern half of the perched aquifer becomes dry within 30 years after the start of remediation.
- (4) Within the site, flow directions are generally toward the north during the entire remediation period. The natural groundwater divide across the center of the site is no longer present, due to the disruption caused by the slurry wall.

Under this remediation pumping scheme, contaminated groundwater in the perched aquifer within the VWP site would not be expected to migrate off-site toward the west, south, or east because of the inward hydraulic gradients along those boundaries. Contaminated groundwater within the perched aquifer may migrate to the north, but it would be slowed by having to cross the low-permeability slurry wall and by the dry conditions in the northern half of the site. Within the slurry walled region, the flow gradients will cause any contaminated groundwater in the perched aquifer to migrate towards the north. Very little migration is expected towards the east or west.

Figures 3-7a, b, and c show the water levels in the saprolite aquifer at one year, 10 years, and 30 years after the start of remediation. The location of the extraction wells in the saprolite aquifer can be seen from the drawdown cones around each extraction well after 1 year. Two important observations can be made from these groundwater elevation maps. First, the drawdown after one year is sufficient to convert the saprolite aquifer within the slurry walled region into an unconfined aquifer. It remains unconfined throughout the 30 year remediation period. Second, inward gradients are created around the entire site within 1 year. These inward gradients remain throughout the remediation period.

Under this remediation pumping scheme, contaminated groundwater within the saprolite aquifer will not be expected to migrate away from the site in any direction.



**FIGURE 3.7a**  
**MODELED POTENTIOMETRIC SURFACE ELEVATIONS**  
**IN SAPPROLITE AQUIFER**  
**AT ONE YEAR AFTER THE START OF REMEDIATION**  
**VIRGINIA WOOD PRESERVING SITE**

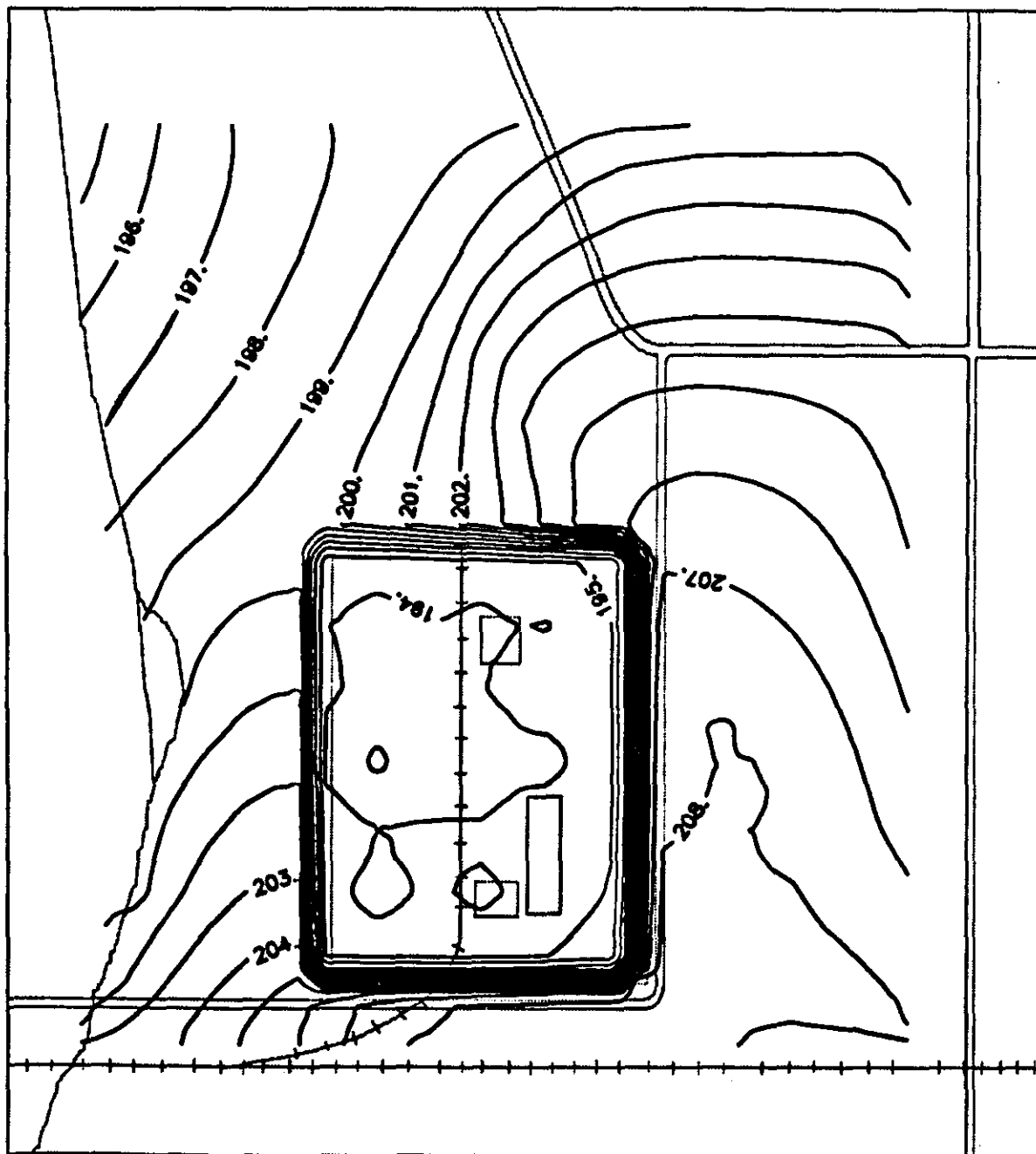


FIGURE 3.7b  
 MODELED POTENTIOMETRIC SURFACE ELEVATIONS  
 IN SAPROLITE AQUIFER  
 AT TEN YEARS AFTER THE START OF REMEDIATION  
 VIRGINIA WOOD PRESERVING SITE

AR304032

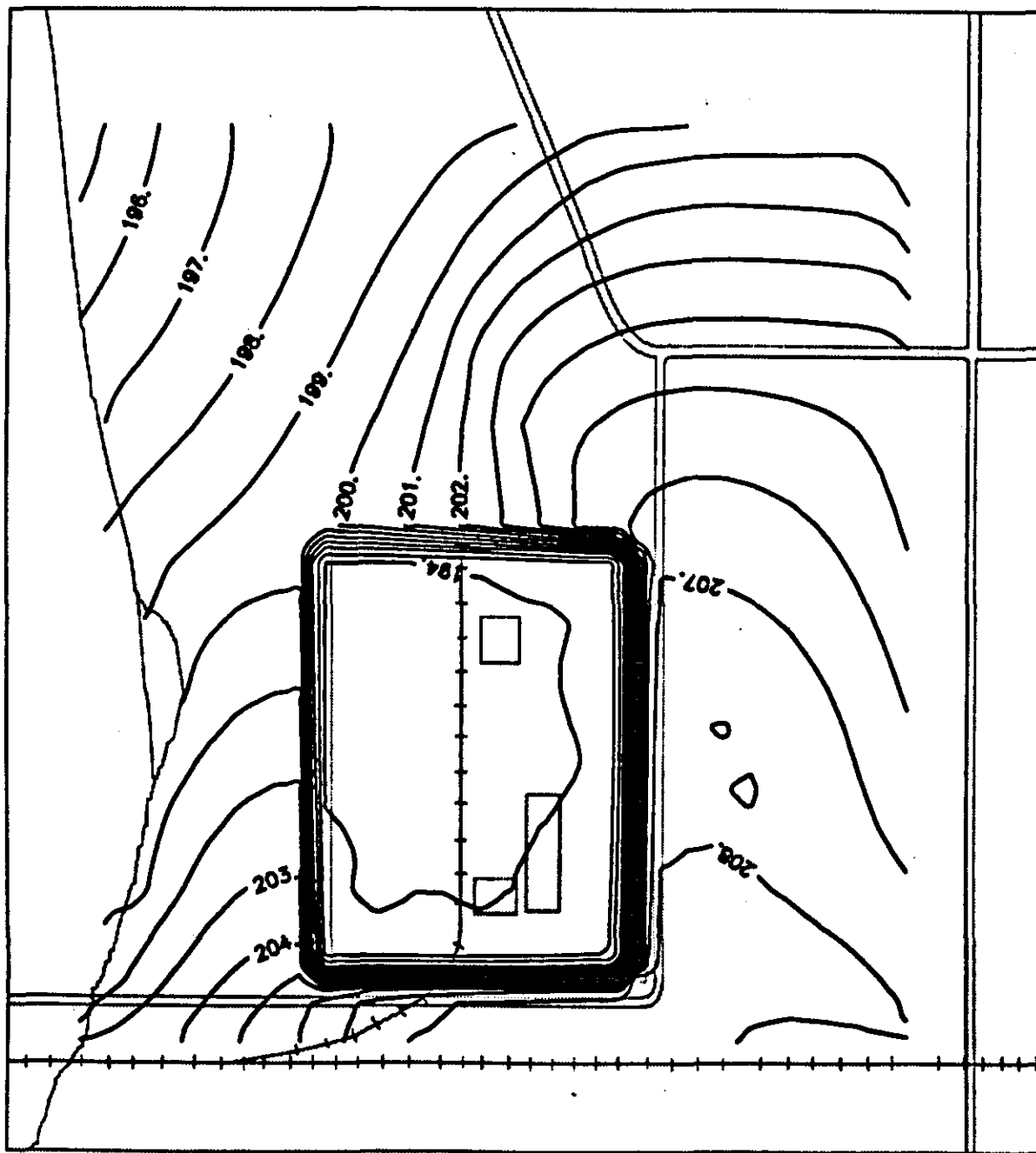


FIGURE 3.7c  
 MODELED POTENTIOMETRIC SURFACE ELEVATIONS  
 IN SAPROLITE AQUIFER  
 AT THIRTY YEARS AFTER THE START OF REMEDIATION  
 VIRGINIA WOOD PRESERVING SITE

Strong inward gradients are created across the slurry wall, with head differences up to 10 feet. The groundwater table within the site is basically flat during most of the remediation period, so the contaminated groundwater within the site will not be compelled to move anywhere.

#### **3.3.2.1.4 Remediation Transport Modeling**

In order to verify the general conclusions obtained from the remediation flow modeling, contaminant transport modeling of the remediation scenarios was undertaken. The mandated remediation measures, described in the introduction, was modeled as remediation Scenario 1. An alternate remediation scenario which simulates leaving the "hot spot" soils in place, was modeled as remediation Scenario 2. The concentration of pentachlorophenol (PCP) throughout the remediation period was predicted for each of the remediation scenarios. From the distribution of PCP, the maximum concentration outside the slurry walled area can be calculated.

The description of the remediation transport model proceeded as follows. First, the possible sources of contamination were reviewed, along with a conceptual discussion of how the contaminants move in the subsurface. Next, the modeling of the sources in the remediation scenarios was described. Third, the parameters used in the transport remediation models were described. Finally, the remediation modeling, results and conclusions were discussed.

##### **3.3.2.1.4.1 Origin and Past Movement of Contaminant Sources**

The contamination originated at the surface within the site and is present in three forms: (1) contaminants sorbed onto soil particles; (2) non aqueous phase liquid contaminants within the pore spaces of the soil matrix; and (3) dissolved contaminants. In general, the organic contaminants of concern, such as PCP, are dense non-aqueous phase liquids (DNAPLs) with low solubilities and high sorption coefficients. The contaminants sorbed onto soil particles have migrated downward through percolation of the small soil particles, and have also migrated laterally due to



stormwater transport of contaminated soil particles. The DNAPLs have migrated downward under the influence of gravity. The dissolved contaminants have migrated laterally within each aquifer under advection and dispersion of the groundwater. The dissolved contaminants also sorb onto soil particles during transport, and move more slowly than the average advective groundwater velocity.

There are three major areas of surficial contamination: the former blowdown sump, the drip pad, and the unlined pond. These three areas contain the highest concentrations of organic contaminants in the soil. The distribution of contaminants in the groundwater of each aquifer is directly affected by the areas of surficial contamination, but many other factors are involved as well. These factors are discussed for the individual aquifers below.

#### Perched Aquifer

The two sources of dissolved contaminants in the perched aquifer are from the DNAPLs and desorption of contaminants from soil particles.

The DNAPLs originated at the surface, and have probably migrated vertically downward to the top of the hardpan. Once at the hardpan, they may migrate down the dip of the hardpan. However, the horizontal migration along the hardpan is probably very limited for two reasons. First, a low in the hardpan elevation in the vicinity of the blowdown sump (Phase II RI, Figure 3-4) prevents horizontal migration. Second, the relative permeabilities of the non-aqueous phase DNAPLs decrease with travel distance. Therefore, the DNAPLs are probably located near to the original source. Solution of the DNAPLs into the dissolved phase occurs slowly, but the DNAPLs provide a constant source of dissolved contaminants.

The contaminated soil particles have moved significant distances from their original sources (see Phase II RI, Figure 4-2, for an example of PCP contamination in surface soils). The desorption of contaminants from the soil particles to the dissolved phase

depends on the distribution coefficient ( $K_d$ ) between the dissolved phase and the sorbed phase. A simple equation which describes the relationship is:

$$\text{Concentration (sorbed phase)} = K_d * \text{concentration (dissolved phase)}$$

The organic contaminants of concern have large  $K_d$  values, so the concentration of the sorbed phase is much larger than the concentration in the dissolved phase. An accurate determination of the  $K_d$  values is difficult, because it depends on the contaminant and the organic content of the soil. It is possible that desorption is not a significant process for this site.

Once the contaminants dissolve in the groundwater, they move in the general direction of groundwater flow. For the perched aquifer, the drip pad and blowdown sump sources are near a groundwater divide, so the contaminants move both northeast and southeast. The plume spreads due to dispersion. The contaminant plume also moves at a rate slower than the advective groundwater velocity, because the contaminants readily sorb onto soil particles in the flow path.

#### Saprolite Aquifer

All of the organic contamination in the saprolite aquifer was transported from the surface, through both the perched aquifer and the hardpan unit. The contaminants may have been transported in either a dissolved phase or as a non-aqueous phase. Transport of contaminated soil particles probably did not occur.

There is evidence that the majority of the contaminants were transported to the saprolite aquifer as a non-aqueous phase. Under this scenario, the DNAPL that has migrated to the hardpan layer continues to migrate through a hole or soft spot in the hardpan to the saprolite aquifer under the force of gravity. The DNAPL, if it is present in sufficient volume to maintain a high relative permeability, then migrates on to the top of the bedrock, where it accumulates. The migration path and the pool both contain DNAPLs, which provide a source for dissolved contaminants. The contaminants dissolve, migrate with the groundwater flux, and then sorb onto uncontaminated soil particles. This process will continue for a long time.

Several lines of evidence support the scenario described above. The saprolite aquifer region directly below the former blowdown sump contains the highest concentrations of contaminants, both in the soil and dissolved. The concentration in the soil at the top of the bedrock (1992 RI, Figure 4-9 for PCP) is much higher than the concentration in the soil at the top of the saprolite (1992 RI, Figure 4-7), indicating a pool of contaminants at the bedrock. The bedrock also contains a topographic depression in this area, which serves to contain the DNAPL.

A smaller center of contamination exists in the saprolite aquifer below the unlined pond. There is no other evidence of a hole in the hardpan at this location, but one must exist if the DNAPLs were able to migrate downward to the saprolite. The concentration in the soil at the top of the bedrock is zero, while the concentration in the soil at the top of the saprolite is high. In this area, the volume of DNAPLs may not be large enough to allow migration as far as the bedrock. The total volume of DNAPLs migrates downward through the pore spaces as continuous stringers. As the stringers migrate, they are stretched and occupy lesser amounts of the pore space. If the stringers become stretched enough so that they occupy less than 15% of the pore space, the relative permeability of DNAPL approaches zero and the downward migration halts. This process appears to have occurred in the saprolite aquifer below the unlined pond. The stringers of DNAPL, while immobile, still provide a source for dissolved contaminants, which will be mobile and move away from the source.

A third area of soil contamination exists in the saprolite at the north-central border of the site. In this area, the hardpan thins to zero or near zero. This area is also a surface drainage. The source of the contaminants may be contaminated soil particles carried by run-off from the site, and then infiltrating through the perched unit and directly into the hardpan. The saprolite groundwater north of the boundary does contain some dissolved contaminants (1994 RDWP, Figure 3-1). The source of these contaminants may be desorption from the contaminated soil particles.

#### **3.3.2.1.4.2 Contaminant Sources for Remediation Models**

The "hot spot" soil areas are the CCA Disposal Area, the Fill Area, and the soil between the surface and the hardpan within 25 feet of the drip pad, the blowdown sump, and the unlined lagoon. The contaminants in the perched aquifer within the "hot spot" areas are a combination of non-aqueous phase and sorbed contaminants. The maps of concentration of contaminants in the soil does not differentiate between these two phases. However, these two phases act differently in their contributions to dissolved contaminants. The non-aqueous phase dissolves slowly but continuously. The sorbed phase can potentially desorb, but this desorption process is not likely if the dissolved concentration is already at a high level due to dissolution of the non-aqueous phase.

The non-aqueous phase in the perched aquifer is also moving into the saprolite aquifer through the holes in the hardpan unit under the force of gravity. The rate of movement is not well known. Also unknown is the amount of non-aqueous phase remaining in the perched aquifer that may migrate into the saprolite aquifer. These unknowns force several assumptions to be made for the simulation of the remediation scenarios.

The initial concentration levels and source locations and concentration levels for remediation Scenario 1 are simulated as follows.

- The initial concentrations of PCP in the perched aquifer are set to their measured dissolved concentration levels, from Figure 4-12 in the 1992 RI, with the exception of the "hot spot" areas. In the "hot spot" areas, the initial concentrations are set to zero, since it is assumed that the excavation and treatment will remove the dissolved contaminants as well as the sorbed contaminants.
- No sources are simulated in the perched aquifer, since the "hot spot" source areas are removed through treatment.
- The initial concentrations in the saprolite aquifer are set to their measured dissolved concentration levels, from Figure 4-14 in the 1992 RI.
- Two contaminant sources, representing non-aqueous phase pools and stringers, are simulated in the saprolite aquifer. One source represents the accumulation of DNAPL on bedrock beneath the former blowdown

sump. The other source represents the stringers of DNAPL below the unlined pond. The concentrations of these sources are set equal to the current measured dissolved concentrations in those areas. The measured dissolved concentration levels are assumed to represent an equilibrium concentration level based on the rate of dissolution and the rate of groundwater flow through those areas. Since the DNAPLs will probably not be removed through extraction during the life of the remediation, the source concentrations are assumed to be constant over the 30 year remediation period.

The initial concentration levels and source locations and concentration levels for remediation Scenario 2 are simulated as follows.

- The initial concentration levels in the perched aquifer are set equal to their measured dissolved concentration levels.
- Two sources are simulated in the perched aquifer, one at the drip pad and sump, and one near the unlined lagoon. These sources represent stringers or subsurface accumulation of DNAPLs on the hardpan. The concentrations of these sources are set equal to the current dissolved concentration levels in those locations. These source concentrations are assumed to be constant over the 30 year remediation scenario. Note that the accumulation of DNAPL below the unlined pond may have already migrated down the slope of the hardpan, towards the east. This migration should be halted by the slurry wall and pumping.
- The initial concentration levels in the saprolite aquifer are set equal to their measured dissolved concentration levels.
- The initial source locations and concentration levels for the saprolite aquifer are the same as in remediation Scenario 1. However, since the perched aquifer was not cleaned up, the volume of this source may increase over time, as more DNAPLs migrate from the perched aquifer to the saprolite aquifer. Because the dissolved concentration levels in the saprolite aquifer are already much higher than the perched aquifer, most of the DNAPLs have probably already migrated. However, as a conservative estimate of the increase in source concentrations, the source concentrations in the saprolite aquifer are simulated to increase during the first 10 years of the remediation period. The concentration at the end of the first 10 years is equal to the maximum current concentration in the saprolite aquifer plus the maximum current concentration in the perched aquifer. This source concentration is then considered constant for the next 20 years.

AR304039

#### **3.3.2.1.4.3 Parameters Used in the Transport Model**

The transport of organic contaminants at the Virginia Wood Preserving Site is simulated using the MT3D model (Zheng, 1992). Because the timing, locations, and concentration of the sources are not well known, it is extremely difficult to calibrate a transport model for this site. Instead, typical transport model parameter values for these soil types and transport distances are used for the remediation simulations: Highly conservative values are used for the retardation and degradation of organic contaminants in the subsurface.

The transport parameters necessary for the remediation scenarios are the initial concentrations, the source concentrations, the porosity and dispersivity values for each aquifer, and the distribution coefficient between the dissolved contaminant and the sorbed contaminants. The initial conditions and source locations are described above. Each layer is assigned a porosity of 0.3, based on the data in the 1992 RI. The longitudinal dispersivity for each aquifer is 10 ft, and the transverse horizontal and transverse vertical dispersivities are 1 ft. These values are based on typical dispersivities for this type of porous media and transport distance.

The distribution coefficient is the most difficult parameter to determine. Several formulas are available to calculate the distribution coefficients, based on parameter such as the octanol-water partition coefficient ( $K_{ow}$ ), the organic carbon partition coefficient ( $K_{oc}$ ), and the fraction of organic carbon in the soil. For PCP, these formulas produce  $K_d$  values ranging between 9 and 308 liters per kilogram. These  $K_d$  values mean that there are between 9 and 308 parts of sorbed contaminant for every part of dissolved contaminant. This range of  $K_d$  values produce a large difference in the transport rate. For transport,  $K_d$  values can be converted to retardation factors. The retardation factor gives an indication of how much the contaminant transport rate is reduced from the groundwater flow rate. Retardation factors for PCP in this aquifer range from 450 to 1540. These retardation factors indicate that if the average groundwater velocity is 1.0 ft/day, the average velocity of the contaminants is between .0022 and 0.00065 ft/day.

Although the  $K_d$  values and retardation factors are quite uncertain, there is a very conservative approach to calculating the maximum concentrations of contaminants outside the slurry walled area during the 30 year remediation period. When no sorption is simulated, the retardation factor is 1.0, and the solute moves with the average velocity of the groundwater. Since the calculated  $K_d$  values produce retardation factors which range from 450 to 1540, using a retardation factor of 1.0 allows the contaminants to move much farther than they would probably move in this aquifer. It is the most conservative approach possible. In addition, the organic contaminants probably degrade naturally in the subsurface. This degradation process has not been modeled, because the rate of natural degradation is unknown. Not including the degradation process is also a conservative approach, because the calculated contaminant concentrations are higher than they probably will be in the aquifer, especially after long periods of time.

#### **3.3.2.1.4.4 Remediation Transport Modeling Results**

The initial conditions, source locations, and transport parameters described above were implemented into the transport model, linked to the transient flow model, and the transient concentrations throughout the site calculated for a the 30 year remediation period. The results of the remediation transport models can be summarized using the isoconcentration contours throughout the two aquifers. Figures 3-8a, b, c, and d are the concentrations of PCP in the perched aquifer for 1 day, 1 year, 10 years, and 30 years after the start of remediation under remediation Scenario 1. Figures 3-9a, b, c, and d are the concentrations of PCP in the saprolite aquifer for 1 day, 1 year, 10 years and 30 years after the start of remediation under Scenario 1. The concentration contours plotted on these maps are the 1.0 ppb, 5.0 ppb, 10 ppb, 100 ppb, 1000 ppb, and 2000 ppb.

The concentrations after 1 day are equal to the current concentrations of PCP (Figure 3-8a). For the perched aquifer, the highest concentrations outside the site occur outside the southern boundary of the site, up to 30 ppb. Significant concentrations are also present outside the eastern boundary of the site, up to 20 ppb. On the northern boundary of the site, the concentrations are only 2 ppb. After one

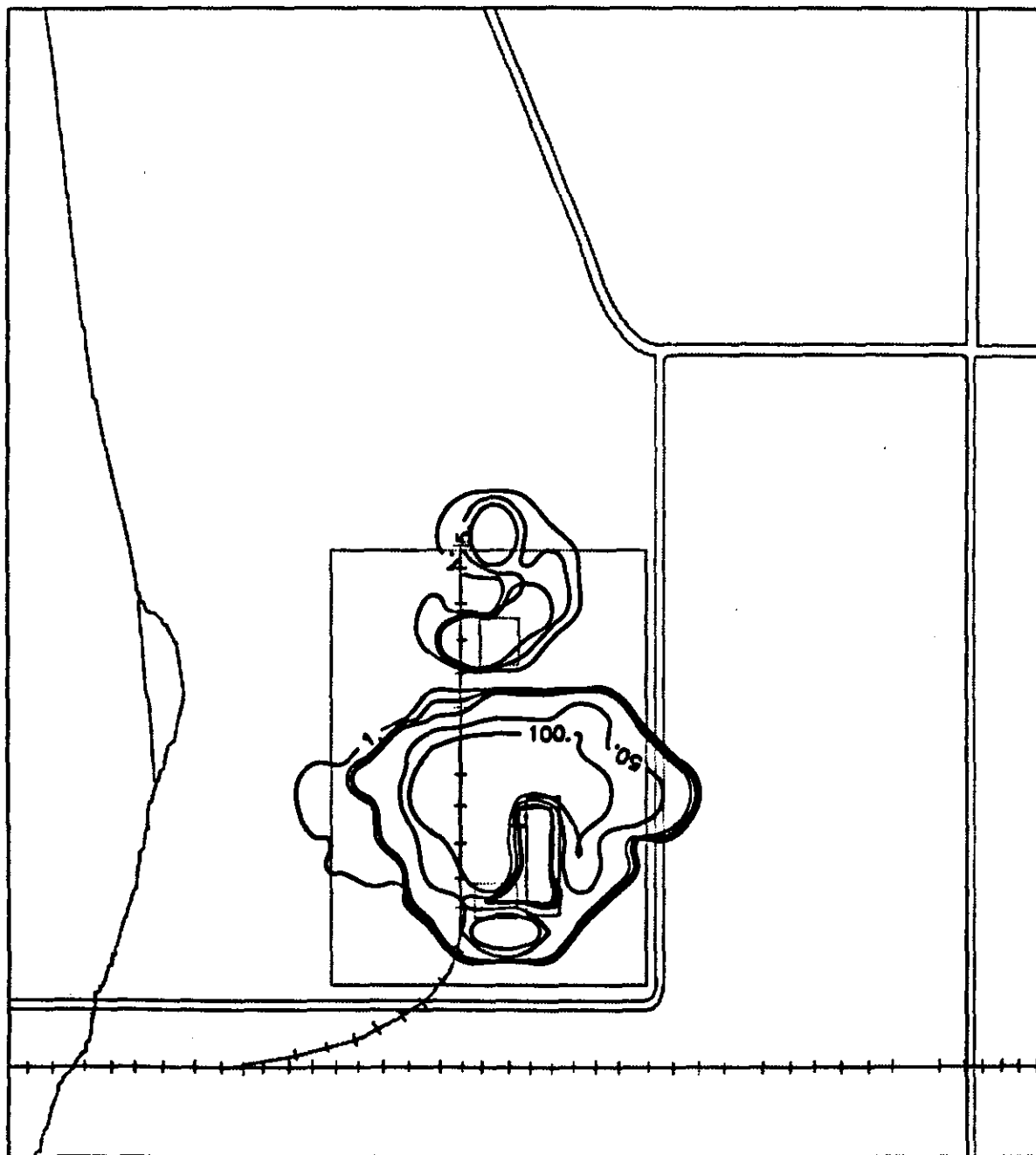
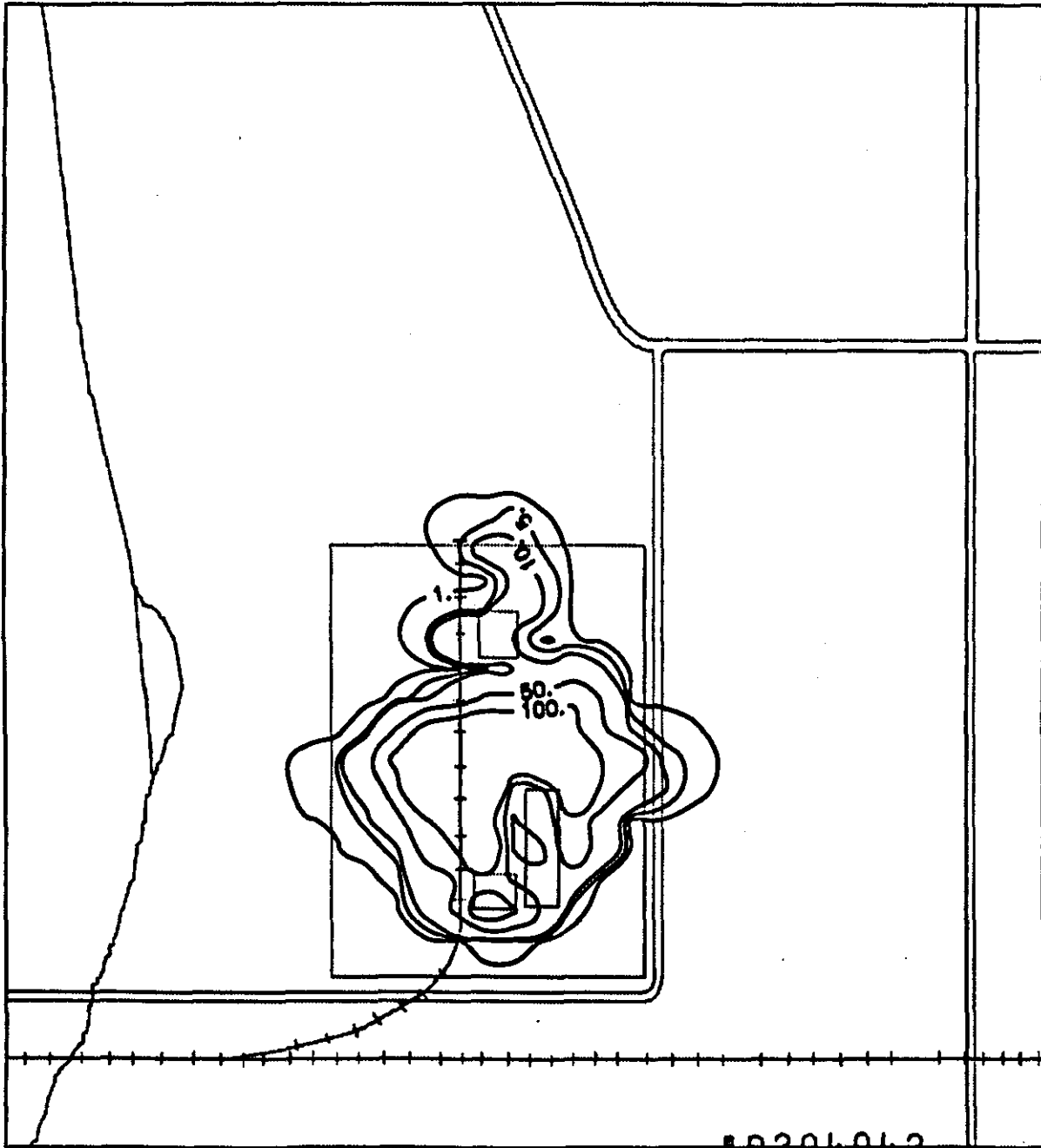


FIGURE 3.8a  
 MODELED PCP DISSOLVED CONCENTRATION CONTOURS  
 IN PERCHED AQUIFER  
 AT THE START OF REMEDIATION  
 REMEDIATION SCENARIO I  
 VIRGINIA WOOD PRESERVING S

AR304042





**FIGURE 3.8b**  
**MODELED PCP DISSOLVED CONCENTRATION CONTOURS**  
**IN PERCHED AQUIFER**  
**AT ONE YEAR AFTER THE START OF REMEDIATION**  
**REMEDICATION SCENARIO I**  
**VIRGINIA WOOD PRESERVING SITE**

AR304043

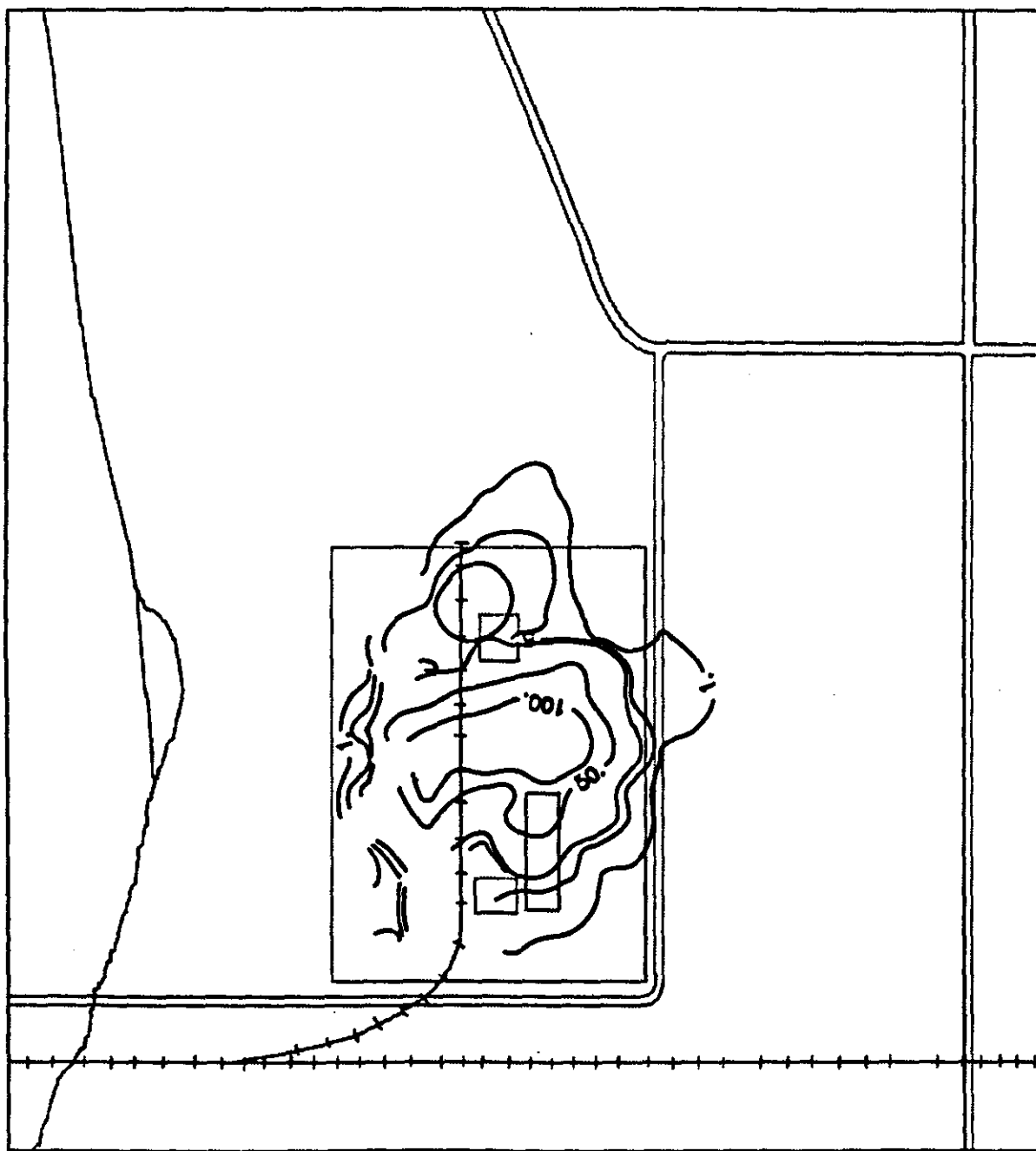
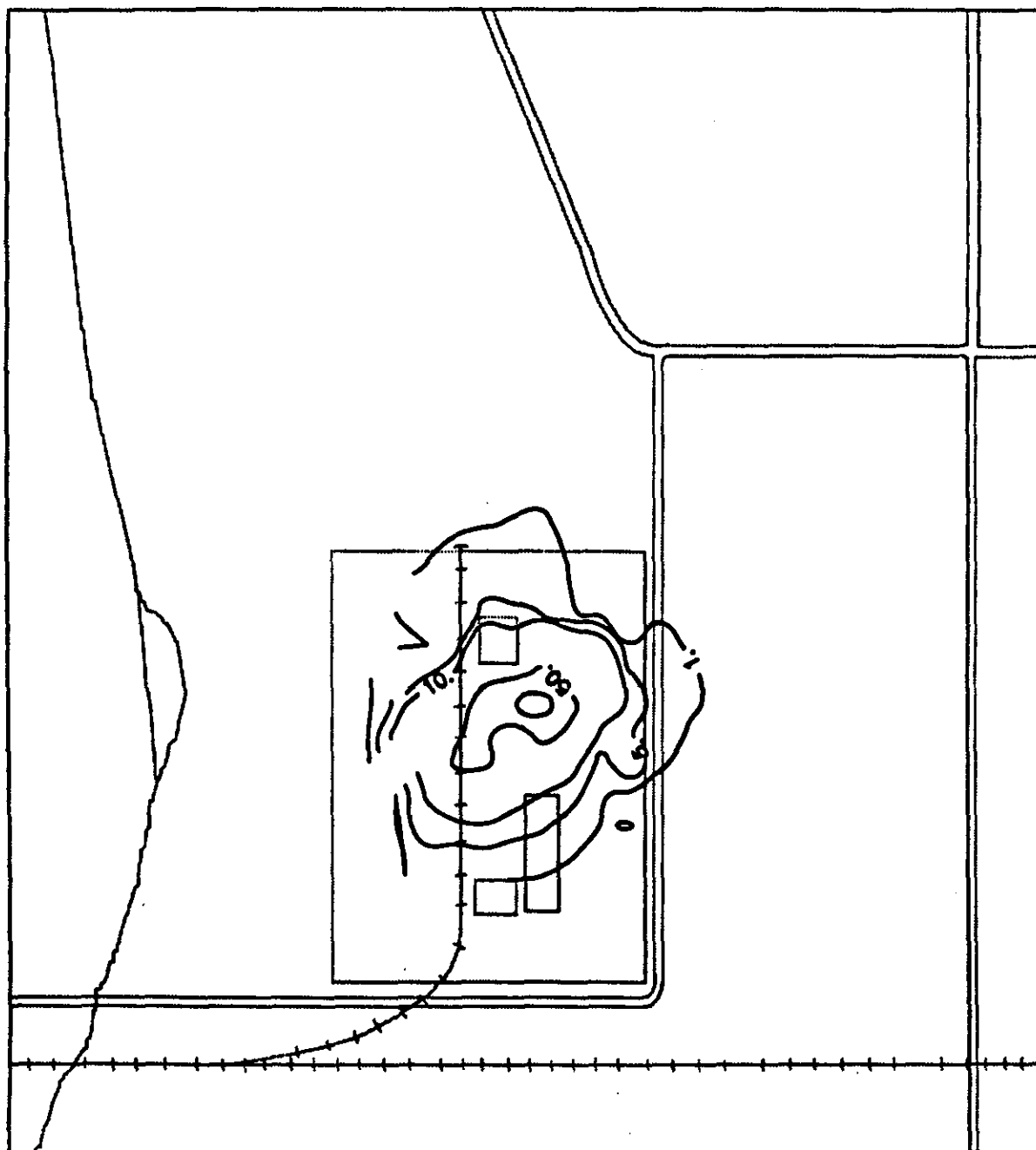


FIGURE 3.8c  
 MODELED PCP DISSOLVED CONCENTRATION CONTOURS  
 IN PERCHED AQUIFER  
 AT TEN YEARS AFTER THE START OF REMEDIATION  
 REMEDIATION SCENARIO 1  
 VIRGINIA WOOD PRESERVING S

AR304044



**FIGURE 3.8d**  
**MODELED PCP DISSOLVED CONCENTRATION CONTOURS**  
**IN PERCHED AQUIFER**  
**AT THIRTY YEARS AFTER THE START OF REMEDIATION**  
**REMEDICATION SCENARIO 1**  
**VIRGINIA WOOD PRESERVING SITE**

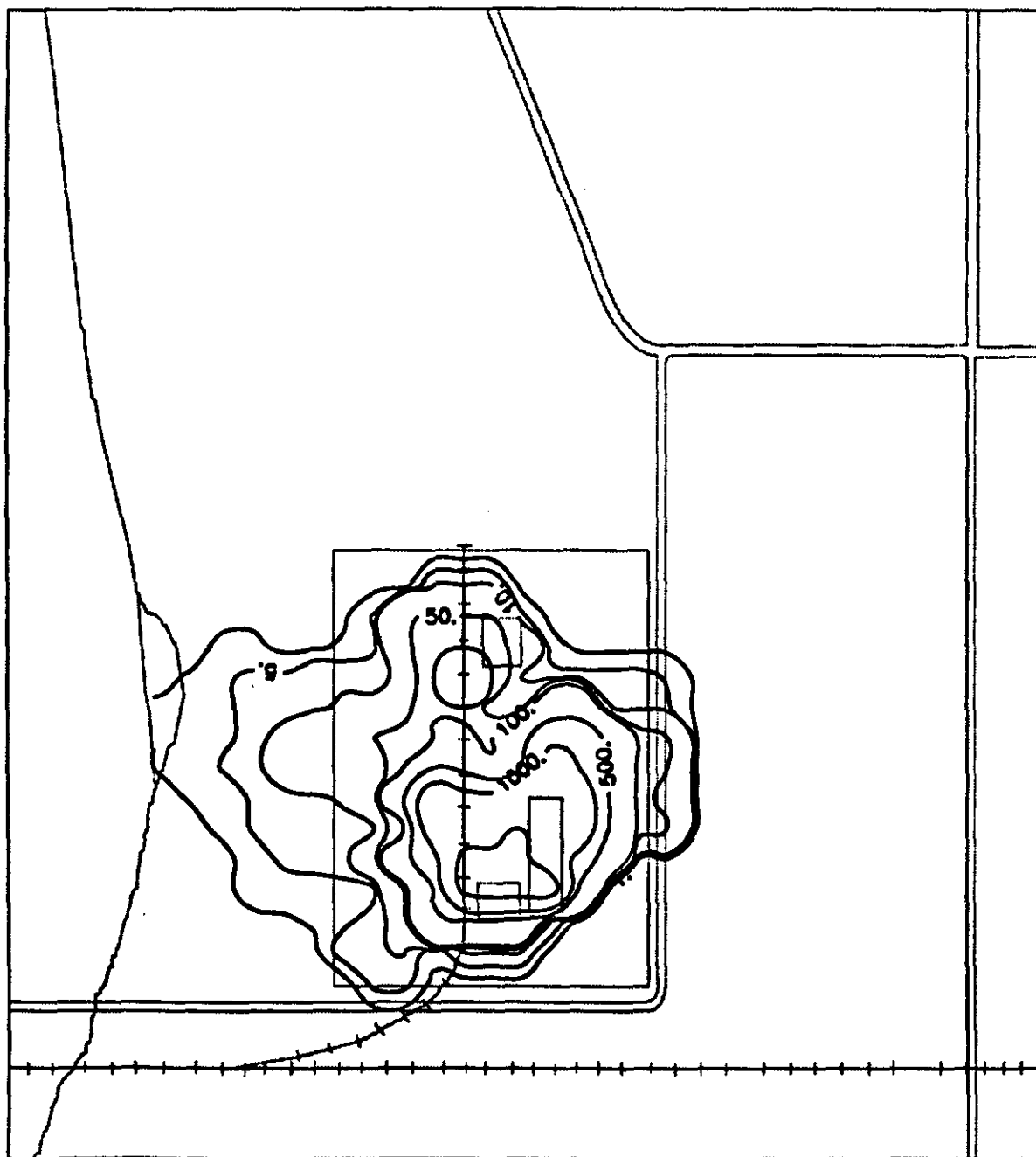


FIGURE 3.9a  
 MODELED PCP DISSOLVED CONCENTRATION CONTOURS  
 IN SAPROLITE AQUIFER  
 AT THE START OF REMEDIATION  
 REMEDIATION SCENARIO 1  
 VIRGINIA WOOD PRESERVING SIX

AR304046

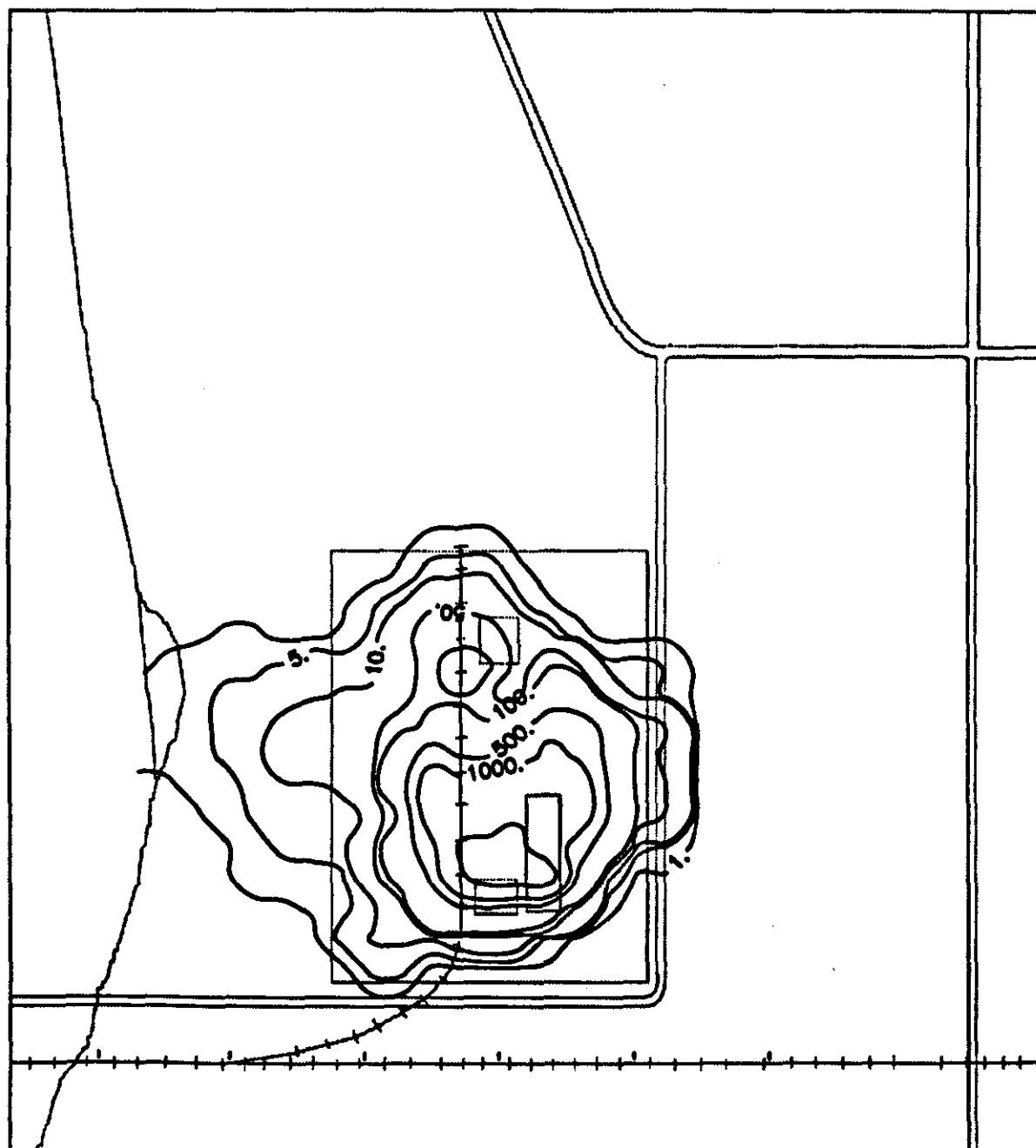
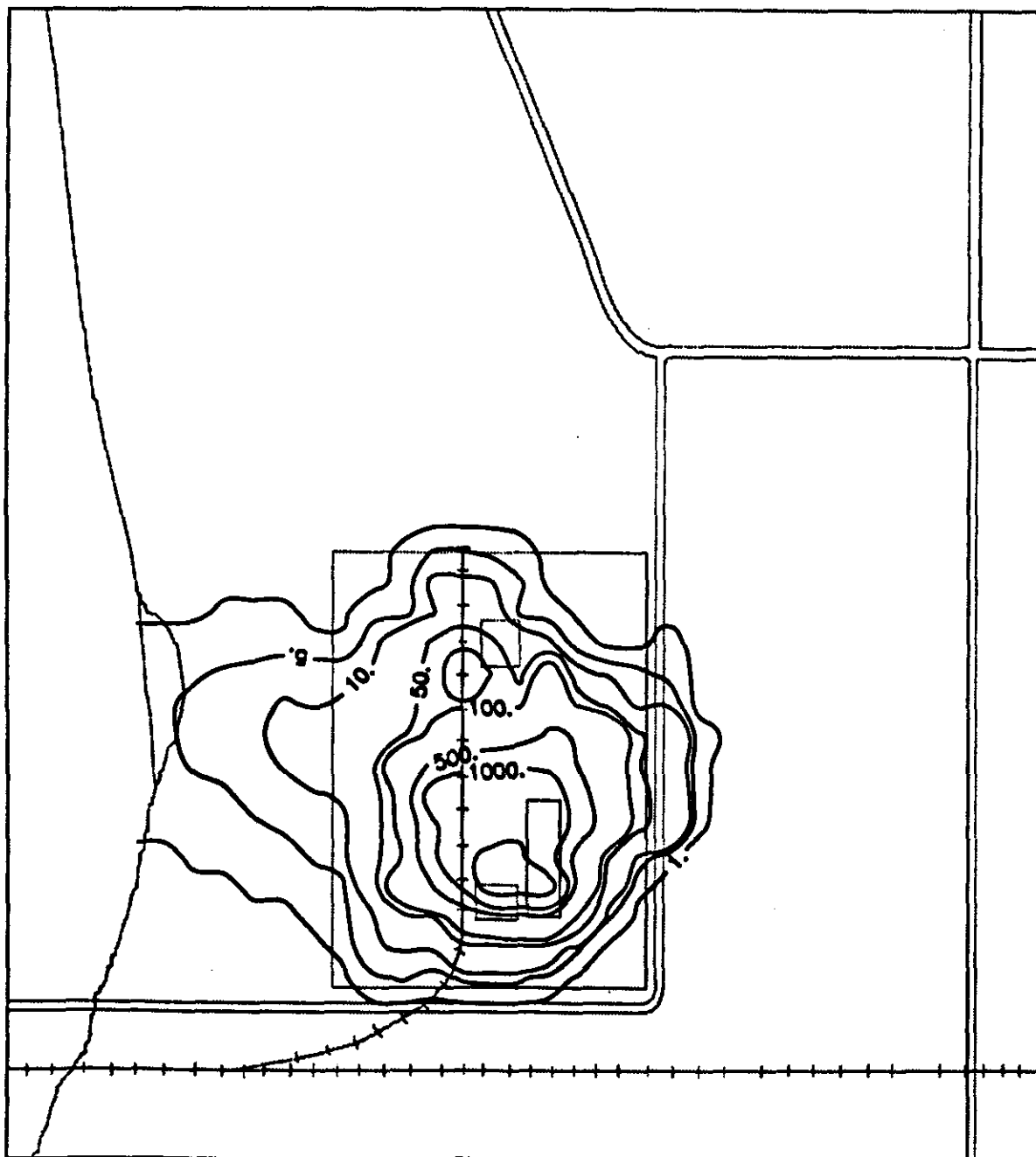


FIGURE 3.9b  
 MODELED PCP DISSOLVED CONCENTRATION CONTOURS  
 IN SAPOLITE AQUIFER  
 AT ONE YEAR AFTER THE START OF REMEDIATION  
 REMEDIATION SCENARIO 1  
 VIRGINIA WOOD PRESERVING SITE



0 200 Feet

Legend

— 5 — Concentration Contour (ppb)

Contours plotted are:

1 ppb  
5 ppb  
10 ppb  
50 ppb  
100 ppb  
500 ppb  
1000 ppb  
2000 ppb

**FIGURE 3.9c**  
**MODELED PCP DISSOLVED CONCENTRATION CONTOURS**  
**IN SAPROOLITE AQUIFER**  
**AT TEN YEARS AFTER THE START OF REMEDIATION**  
**REMEDICATION SCENARIO 1**  
**VIRGINIA WOOD PRESERVING SITE**

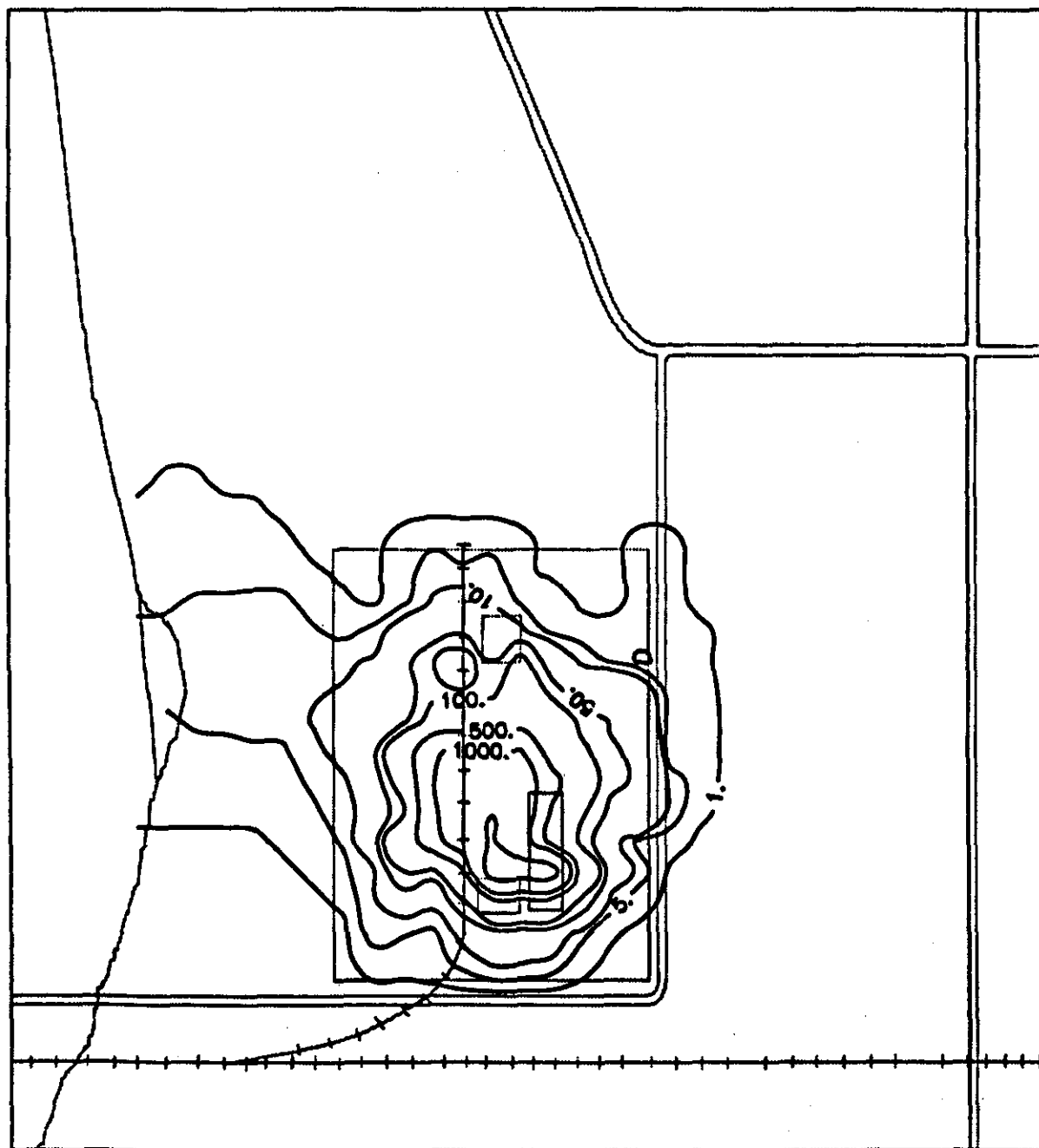


FIGURE 3.9d  
 MODELED PCP DISSOLVED CONCENTRATION CONTOURS  
 IN SAPROLITE AQUIFER  
 AT THIRTY YEARS AFTER THE START OF REMEDIATION  
 REMEDIATION SCENARIO 1  
 VIRGINIA WOOD PRESERVING SITE

AR304049

year (Figure 3-8b), the contaminated area outside the site has remained fairly constant, and the concentrations have decreased slightly. After 10 years (Figure 3-8c), parts of the perched aquifer have dried out. The area of contaminated groundwater has increased slightly due to transport around the slurry wall and dispersion, but the concentrations outside the slurry wall have decreased significantly. After 30 years (Figure 3-8d), the area of contaminated groundwater has remained relatively constant, and the concentrations have decreased even more. The concentrations of PCP decrease over time because no new contamination is crossing the slurry wall from the inside of the site. The existing mass of dissolved contaminants spread out over a larger area due to advection and dispersion, and their concentrations are reduced.

In the saprolite aquifer, the concentrations after day 1 are equal to the current concentrations of PCP (Figure 3-9a). Note that the initial concentrations of PCP within the site are much larger in the saprolite aquifer than in the perched aquifer. The highest concentrations outside the slurry wall are again on the south side of the site, up to 50 ppb. The largest area of contaminated groundwater is found on the north side of the site, since the dominant flow direction within the saprolite aquifer is towards the north. The concentrations outside the slurry wall to the north are up to 15 ppb. Over time, the contaminated water around the south side of the slurry wall moves toward the east, as the groundwater flows around the slurry wall. The concentrations on the south side decrease, as dispersion processes spread the existing contaminant over a larger volume. On the north side, the contaminated groundwater already in the saprolite moves toward the creek over time, and the area of contaminated groundwater increases due to advection and dispersion. The concentrations outside the northern side also decrease over time, because no new mass is added from inside the site.

The maps of concentration contours for remediation Scenario 1 can be compared to the maps for remediation Scenario 2 to determine the benefit gained by excavating the "hot spot" soils. Figures 3-10a, b, c, and d are the concentrations of PCP in the perched aquifer for 1 day, 1 year, 10 years, and 30 years after the start of remediation under remediation Scenario 2. Figures 3-11a, b, c, and d are the concentrations of PCP in the saprolite aquifer for 1 day, 1 year, 10 years and 30 years after the start of remediation under Scenario 2.



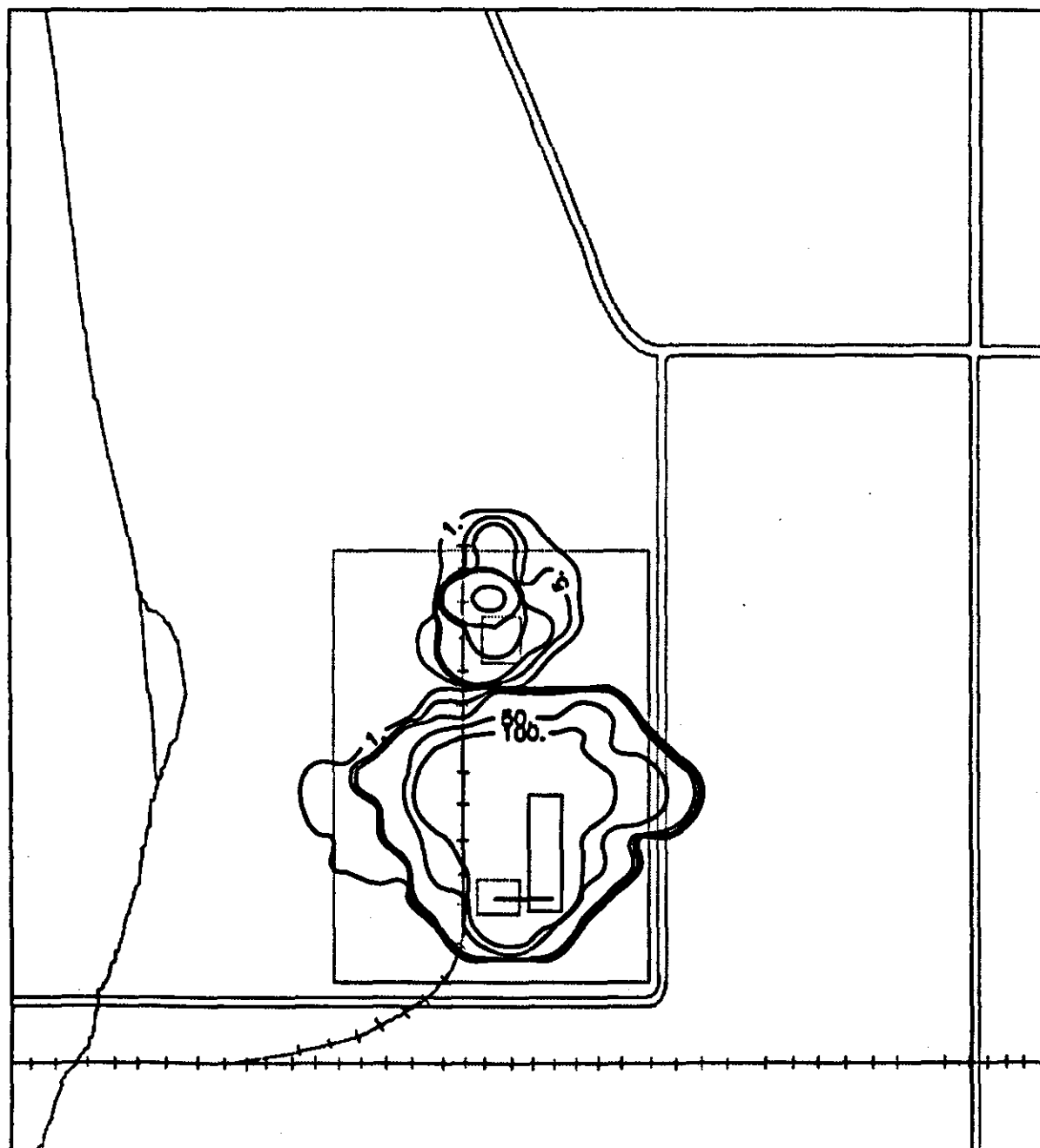


FIGURE 3.10a  
 MODELED PCP DISSOLVED CONCENTRATION CONTOURS  
 IN PERCHED AQUIFER  
 AT THE START OF REMEDIATION  
 REMEDIATION SCENARIO 2  
 VIRGINIA WOOD PRESERVING SITE

AR304051

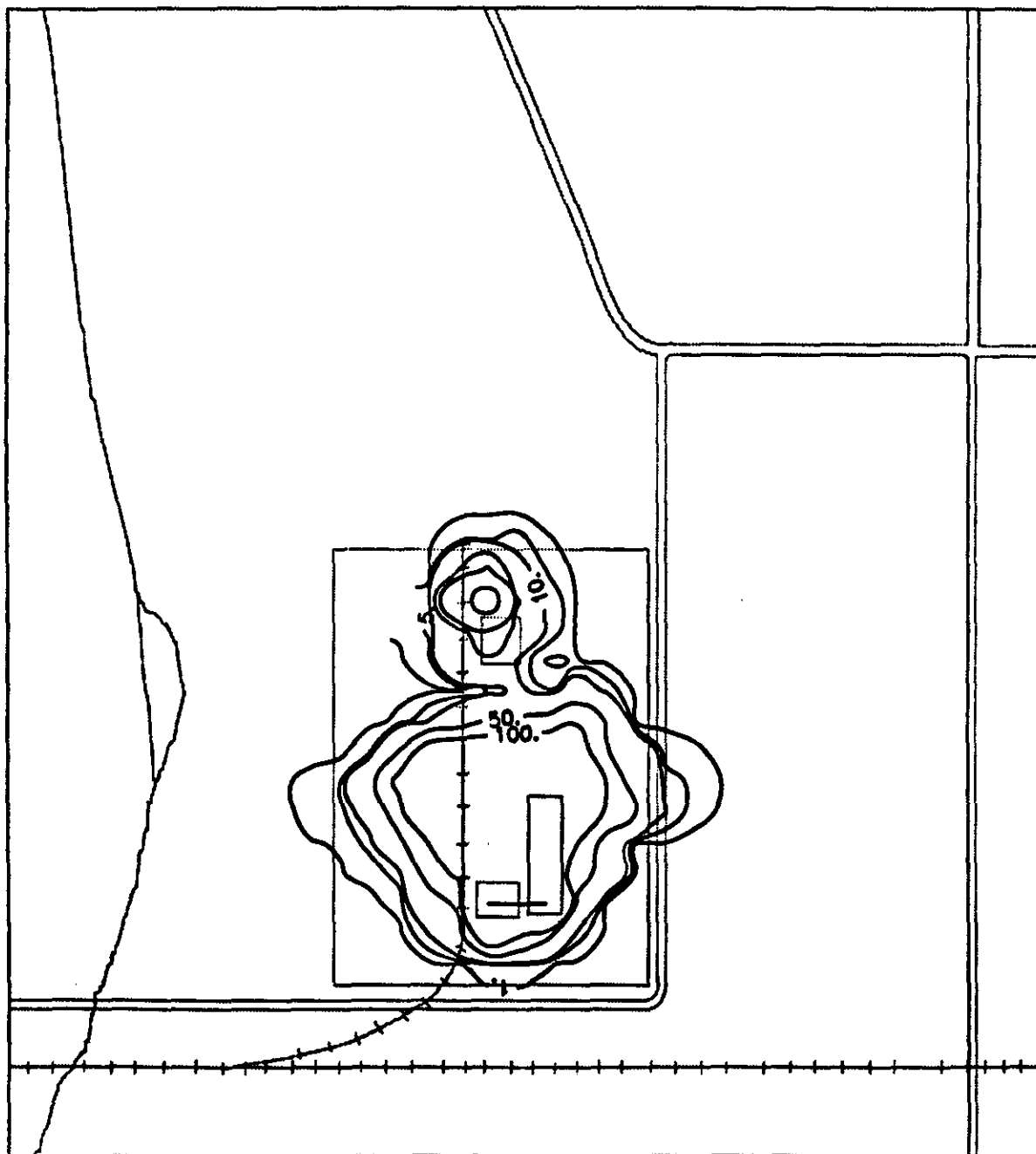


FIGURE 3.10b  
 MODELED PCP DISSOLVED CONCENTRATION CONTOURS  
 IN PERCHED AQUIFER  
 AT ONE YEAR AFTER THE START OF REMEDIATION  
 REMEDIATION SCENARIO 2  
 VIRGINIA WOOD PRESERVING S

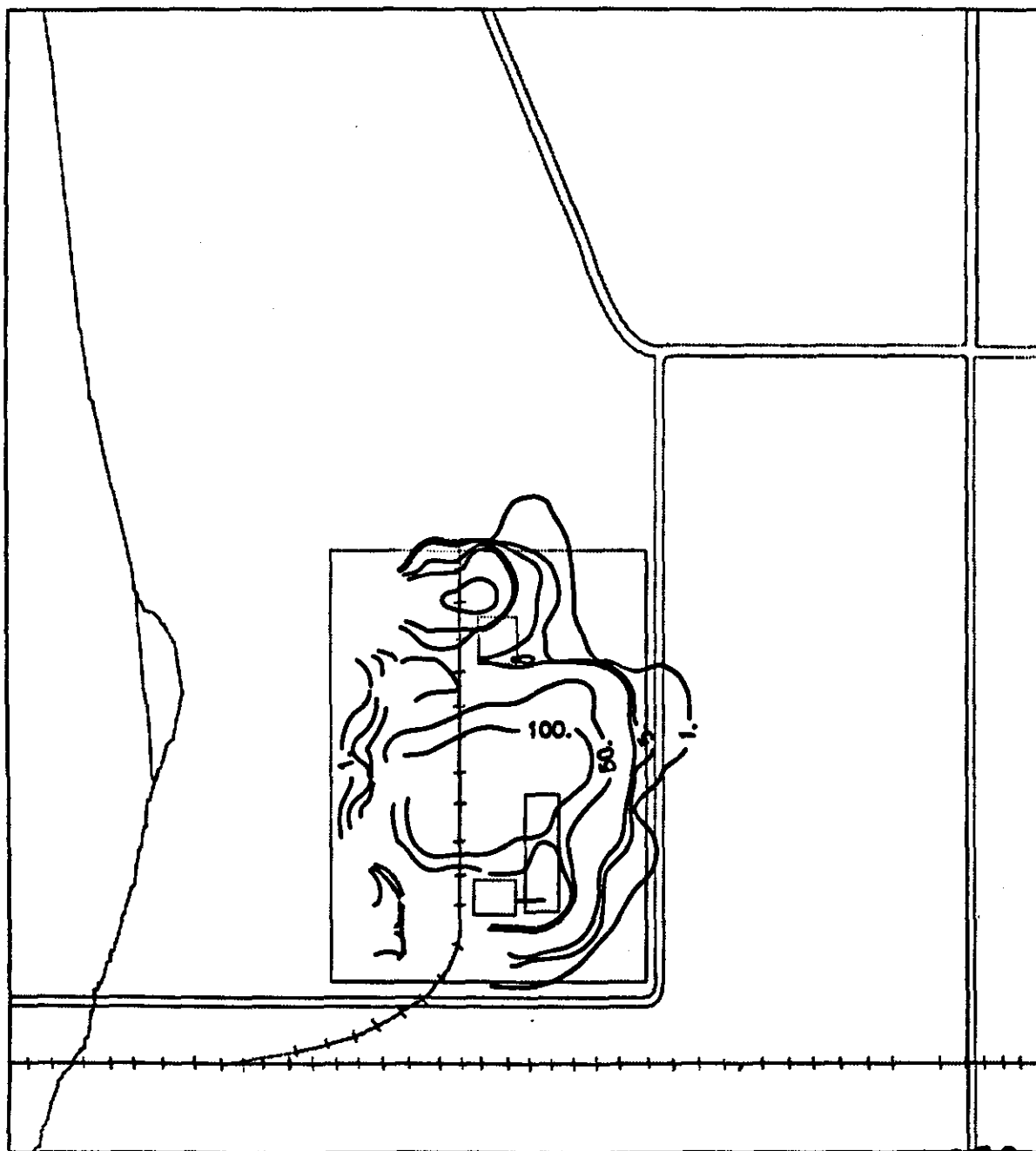


FIGURE 3.10c  
 MODELED PCP DISSOLVED CONCENTRATION CONTOURS  
 IN PERCHED AQUIFER  
 AT TEN YEARS AFTER THE START OF REMEDIATION  
 REMEDIATION SCENARIO 2  
 VIRGINIA WOOD PRESERVING SITE

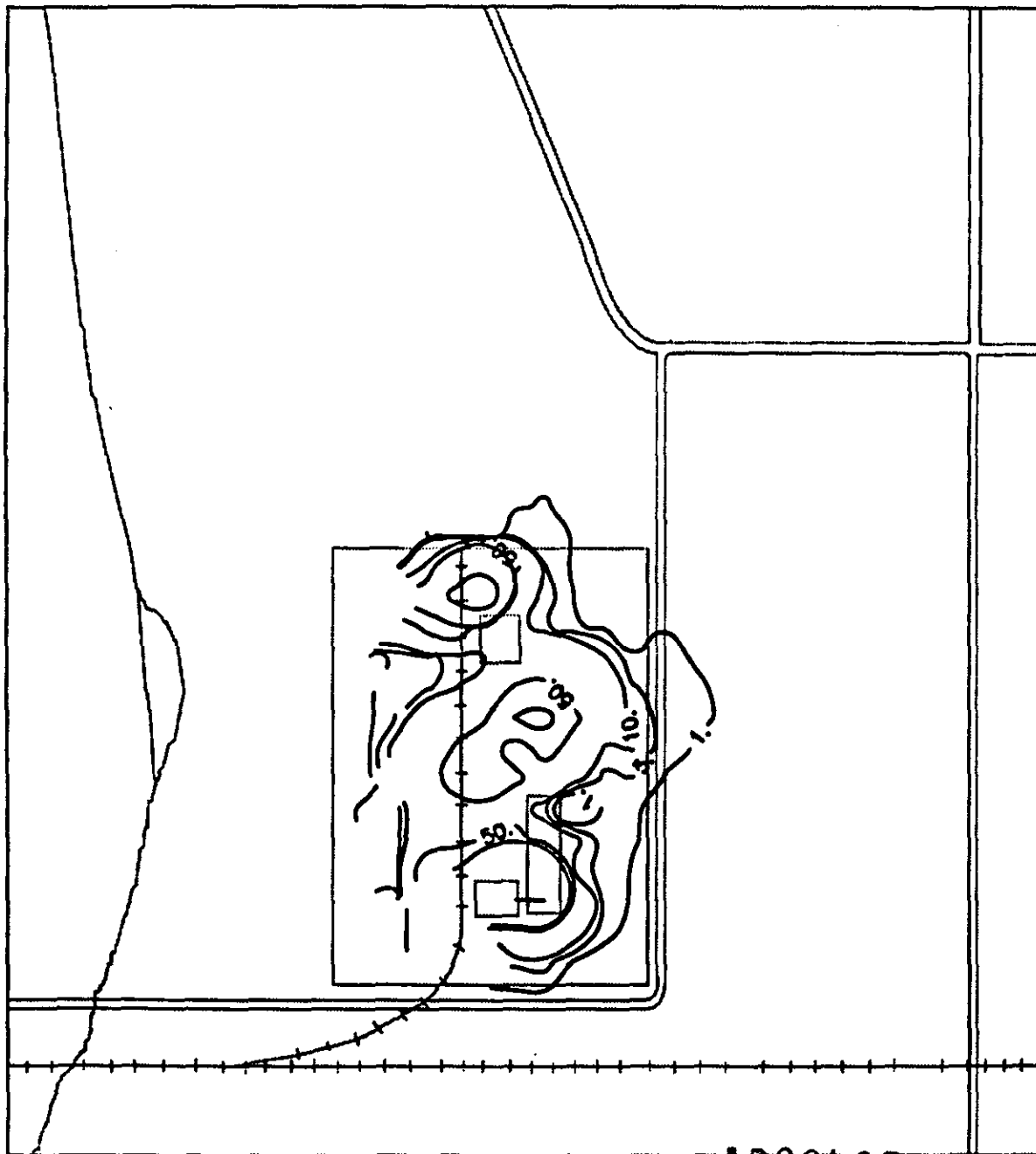


FIGURE 3.10d  
 MODELED PCP DISSOLVED CONCENTRATION CONTOURS  
 IN PERCHED AQUIFER  
 AT THIRTY YEARS AFTER THE START OF REMEDIATION  
 REMEDIATION SCENARIO 2  
 VIRGINIA WOOD PRESERVING

AR304054

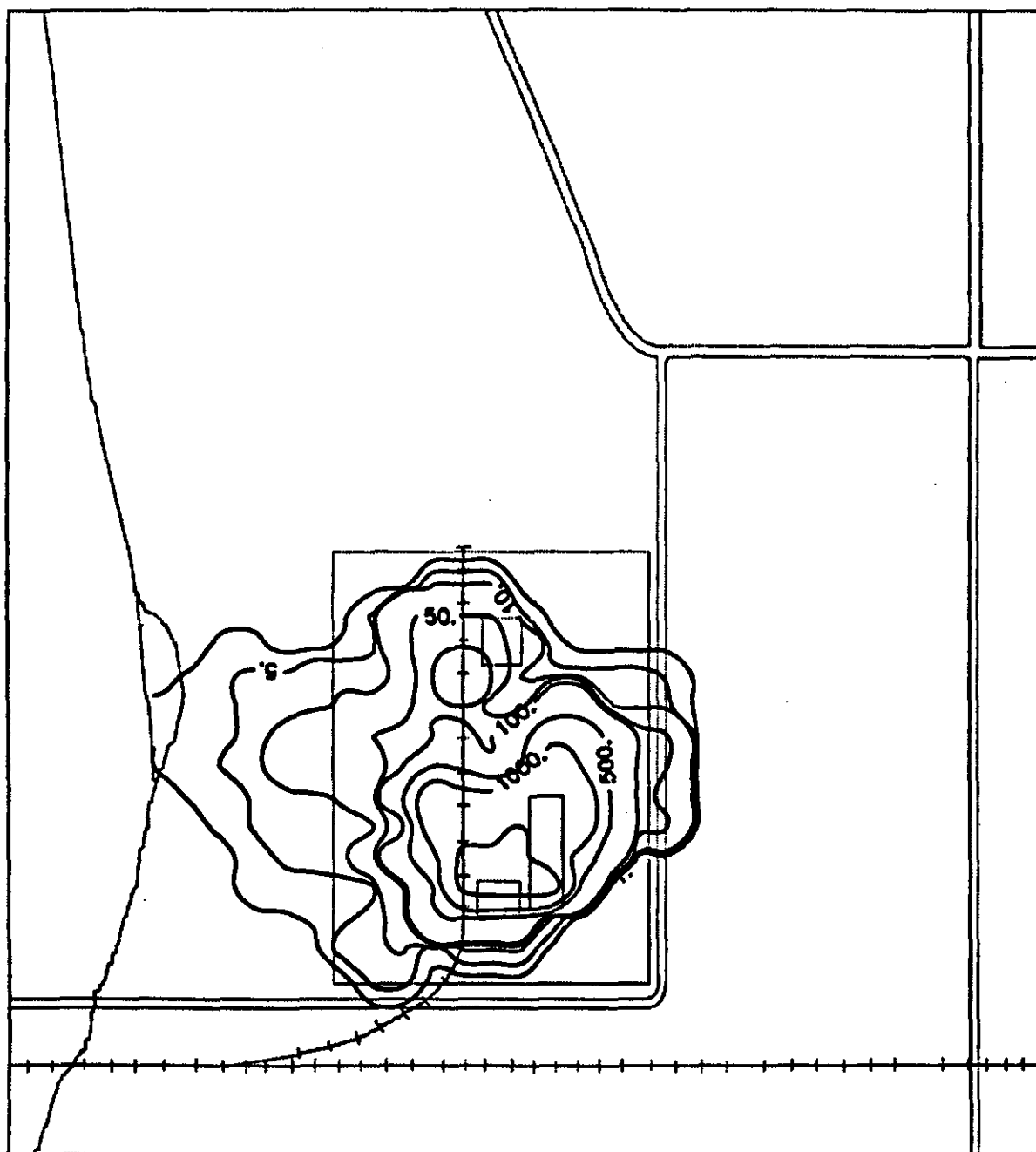


FIGURE 3.11a  
MODELED PCP DISSOLVED CONCENTRATION CONTOURS  
IN SAPROLITE AQUIFER  
AT THE START OF REMEDIATION  
REMEDATION SCENARIO 2  
VIRGINIA WOOD PRESERVING SITE



0 200 Feet

Legend

--- 5 --- Concentration Contour (ppb)

Contours plotted are:

1 ppb  
5 ppb  
10 ppb  
50 ppb  
100 ppb  
500 ppb  
1000 ppb  
2000 ppb

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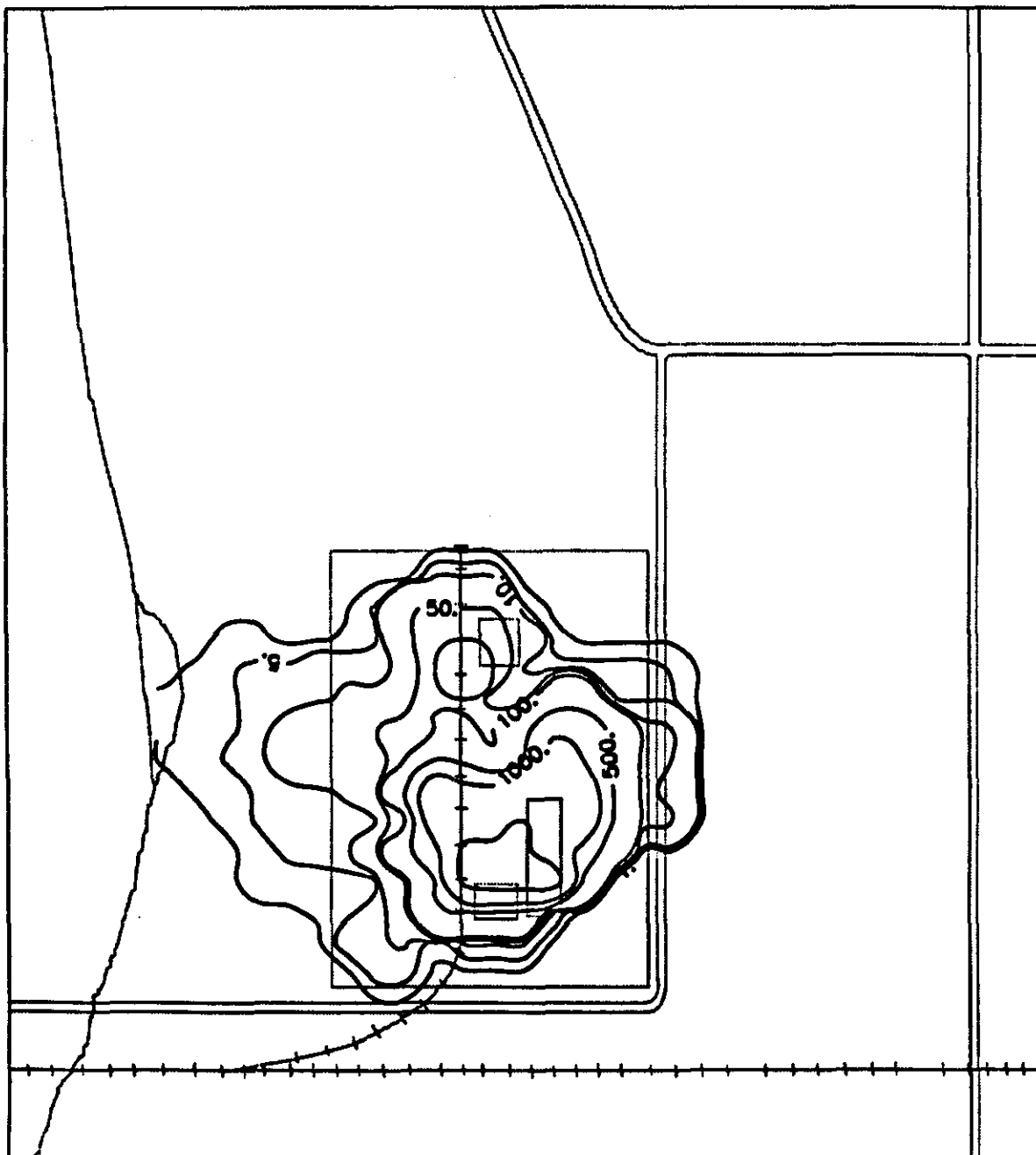


FIGURE 3.11b  
 MODELED PCP DISSOLVED CONCENTRATION CONTOURS  
 IN SAPROLITE AQUIFER  
 AT ONE YEAR AFTER THE START OF REMEDIATION  
 REMEDIATION SCENARIO 2  
 VIRGINIA WOOD PRESERVING S

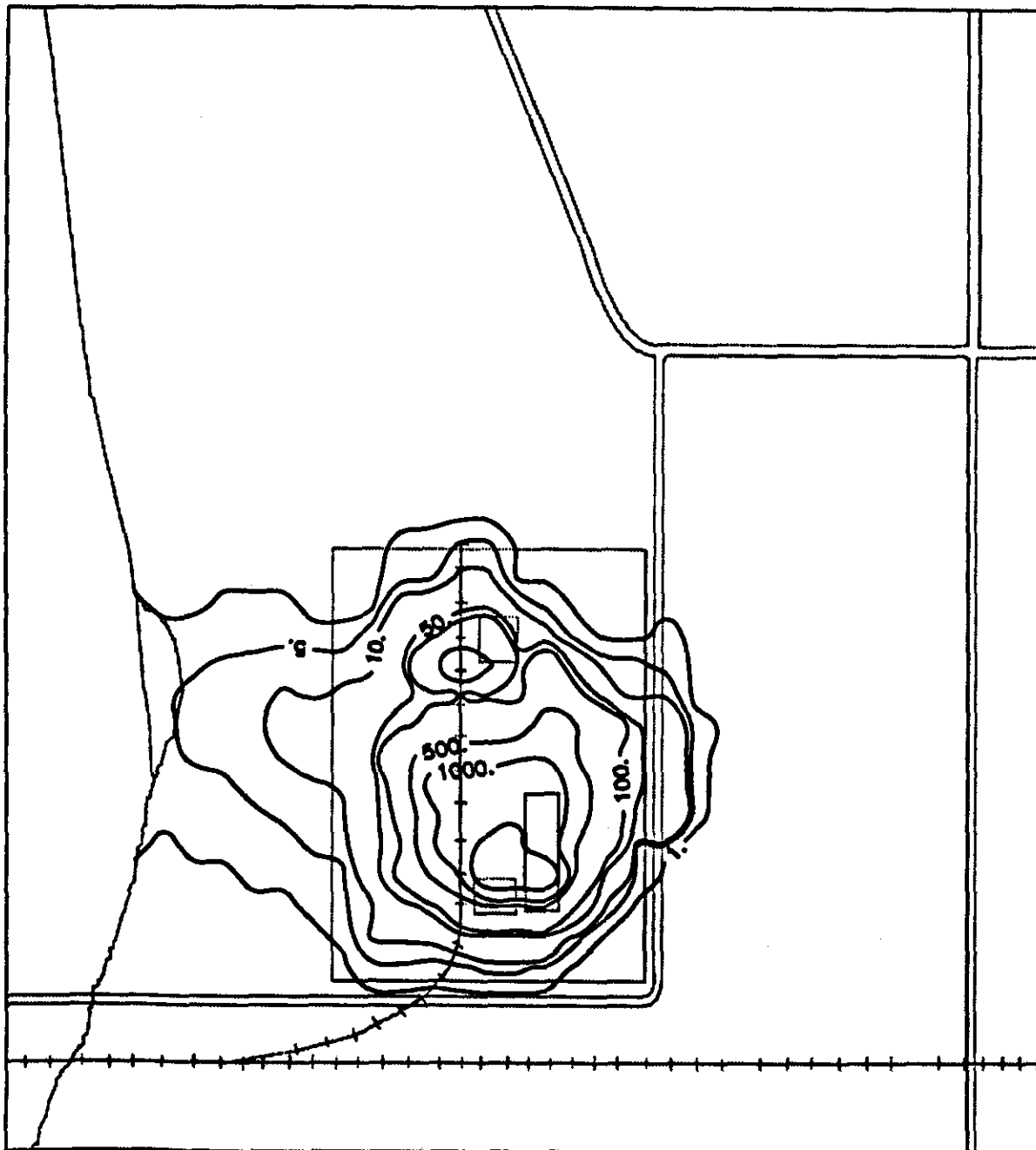


FIGURE 3.11c  
 MODELED PCP DISSOLVED CONCENTRATION CONTOURS  
 IN SAPROLYTE AQUIFER  
 AT TEN YEARS AFTER THE START OF REMEDIATION  
 REMEDIATION SCENARIO 2  
 VIRGINIA WOOD PRESERVING SITE

Legend

— 5 — Concentration Contour (ppb)

Contours plotted are:

- 1 ppb
- 5 ppb
- 10 ppb
- 50 ppb
- 100 ppb
- 500 ppb
- 1000 ppb
- 2000 ppb

0 200 Feet

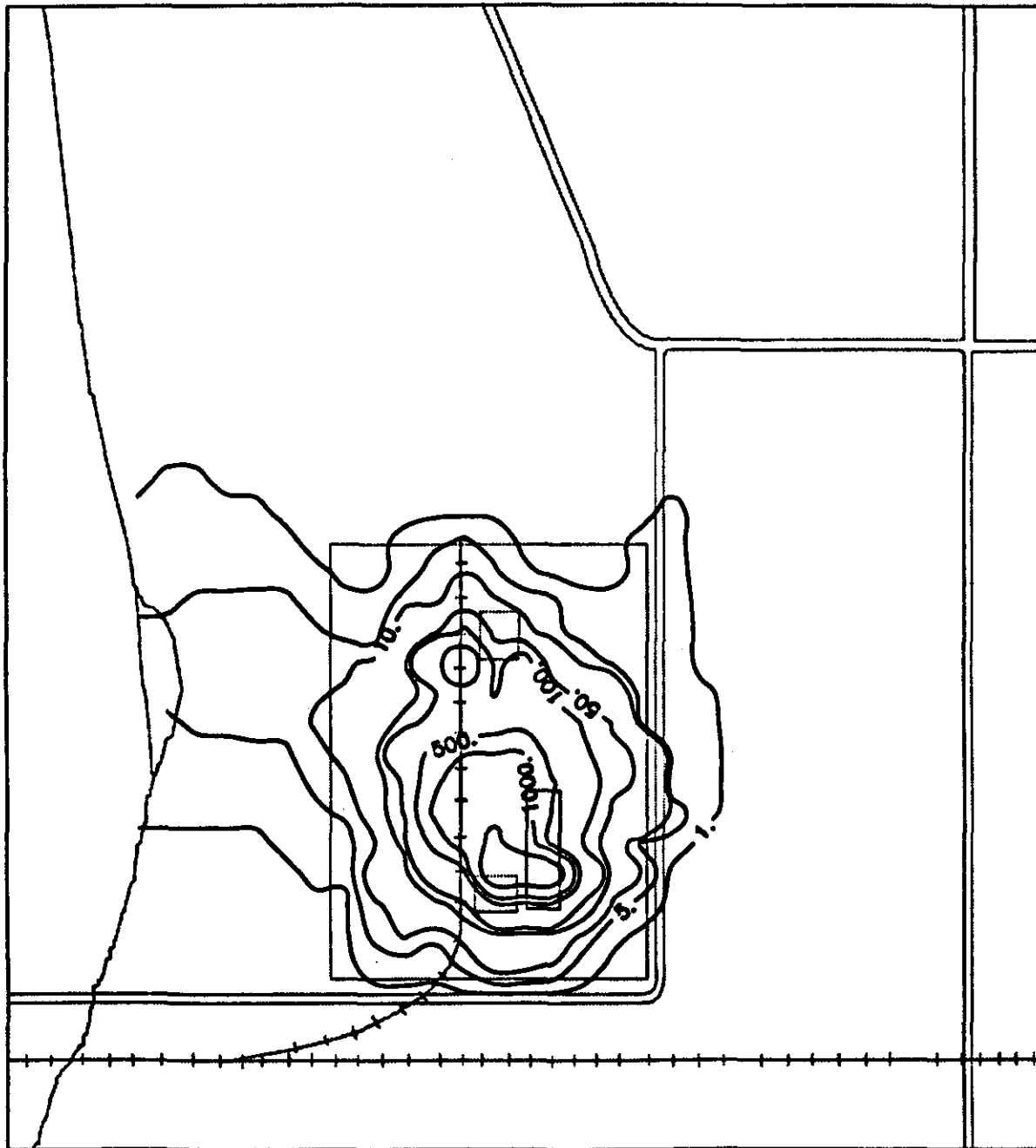


FIGURE 3.11d  
 MODELED PCP DISSOLVED CONCENTRATION CONTOURS  
 IN SAPROLITE AQUIFER  
 AT THIRTY YEARS AFTER THE START OF REMEDIATION  
 REMEDIATION SCENARIO 2,  
 VIRGINIA WOOD PRESERVING S.



The main differences in the perched aquifer between Scenario 1 and Scenario 2 are the presence of sources within the site area. On the southern boundary, the concentrations for the two scenarios are very similar. Outside the eastern boundary, the concentrations for remediation Scenarios 1 and 2 are initially the same, but over time the concentrations in Scenario 2 decay at a slower rate than the concentrations in Scenario 1.

In the saprolite aquifer, very little difference exists between remediation Scenarios 1 and 2. For remediation Scenario 2, the source concentration inside the site increase over time. However, since these sources are not located close to the site boundaries, and inward gradients are maintained during the entire remediation period, the concentrations outside the site are basically unchanged between remediation Scenarios 1 and 2. Removing the "hot spot" soils in the perched aquifer provides almost no benefit to the concentrations or area of contaminated water in the saprolite.

These remediation scenarios are conducted under very conservative assumptions, including no retardation and no biodegradation. These conservative assumptions allows the maximum differences between the two remediation scenarios to be illustrated. If retardation were included, the transport rates would be significantly slower, and the differences between the two remediation scenarios would be significantly smaller.

#### **3.3.2.1.5 Summary and Conclusions**

The fate and transport of organic contaminants at the Virginia Wood Preserving Site has been modeled using a three dimensional groundwater flow and transport model. The concentrations of PCP throughout the site have been calculated under two different remediation scenarios. The two remediation scenarios were developed in order to determine whether removing the "hot spots" of soil contamination in the perched aquifer results in lower contamination levels outside the slurry wall during the remediation period. The PCP concentrations were modeled using highly conservative assumptions for initial concentrations and transport parameters.

From the remediation flow model, inward gradients are created by the remediation measures across all boundaries of the site in the saprolite aquifer, and across all but the northern boundary in the perched aquifer. Contaminated groundwater at the site within the perched aquifer may only migrate towards the north after remediation. The rate of migration is slowed significantly by the low-permeability slurry wall and dry conditions in the northern half of the site during remediation. Within the slurry wall region, the flow gradients will cause any contaminated groundwater in the perched aquifer to migrate towards the north. Contaminated groundwater within the saprolite aquifer is not be expected to migrate away from the site in any direction, due to the strong inward hydraulic gradients.

The remediation transport modeling allows maps of concentration contours for PCP to be constructed throughout the remediation period. These maps can be compared for remediation Scenarios 1 and 2 to determine whether there are any significant differences in concentrations outside the site boundaries.

In the perched aquifer, both remediation scenarios have the highest concentrations outside the site boundary at the beginning of the remediation period, and these concentrations decay over time. The inward groundwater gradients do not allow any contaminants to advectively cross the slurry wall towards the outside. The mass of contaminants currently outside the site spread over a larger area due to advection and dispersion, and therefore decrease in concentration.

In the saprolite aquifer, the highest concentrations outside the site under both remediation scenarios occurs again at the beginning of the remediation period. The highest concentrations outside the slurry wall are on the south side of the site. However, the largest area of contaminated groundwater is found on the north side of the site, since the dominant flow direction within the saprolite aquifer is towards the north. Over time, the contaminated water around the south side moves toward the east, as the groundwater flows around the slurry wall. On the north side, the contaminated groundwater already outside the site moves toward the creek over time, and the area of contaminated groundwater increases due to advection and dispersion. The concentrations of PCP decrease over time, because no new contamination is crossing the slurry wall from the inside of the site.

Very little difference exists between the two remediation scenarios for concentrations within the saprolite aquifer during the 30 year remediation period. Most of the DNAPLs have already migrated as far as they will go into the saprolite aquifer, so removing the DNAPLs in the perched aquifer will not significantly affect the source concentrations in the saprolite. For remediation Scenario 2, a very conservative approach was to assume that the source concentrations inside the site increase over time. However, since these sources are not located close to the site boundaries, and inward gradients are maintained during the entire remediation period, the concentrations outside the site are basically unchanged between remediation scenarios 1 and 2. Removing the "hot spot" soils in the perched aquifer provides almost no benefit to the concentrations or area of contaminated water in the saprolite.

### **3.3.2.2 Analysis of Risk**

#### **3.3.2.1.1 Selection of COCs for the VE Risk Analysis**

Two indicator constituents were selected for evaluation in the VE Risk Analysis. The indicators were chosen from the list of constituents which were considered in the selection the LTDD remedial action in the Record of Decision (ROD) for the site (USEPA 1993). The list of constituents includes pentachlorophenol (PCP), carcinogenic PAHs (PAH<sub>s</sub>), and arsenic. The primary consideration in the selection of LTDD as a remedy was the reduction of potential carcinogenic risks associated with these constituents. The levels of arsenic in soil and groundwater outside the slurry wall will not be impacted by LTDD, and will remain constant under each remediation scenario. Since relative risks for arsenic exposure in each scenario would not be altered by LTDD treatment, arsenic was eliminated from evaluation in the comparative risk analysis. PCP and PAH<sub>s</sub> concentrations in soil will be reduced by LTDD and were the primary constituents of concern in the decision process leading to the ROD, and were thus selected for evaluation in the VE Risk Analysis.

The structure and activity relationships of the PAH<sub>s</sub> are similar and these constituents are often evaluated as a single group. Further, as carcinogenic toxicity criteria are not available for all the PAH<sub>s</sub>, risk evaluation is usually based on the criteria developed for

benzo(a)pyrene (B(a)P). Thus, rather than conducting groundwater modeling for all the PAH<sub>s</sub>, a single PAH<sub>s</sub> was chosen based on the greatest mobility and toxicity criteria, as described below.

#### Comparative Mobility of PAH<sub>s</sub>

The relative mobility of the PAH<sub>s</sub> was compared using the organic-carbon partition coefficient,  $K_{oc}$ , and the octanol-water partition coefficient,  $K_{ow}$ .  $K_{oc}$  is an indicator of a constituent's tendency to adsorb to organic matter in soil. This adsorption of nonpolar organic constituents is treated as an equilibrium-partitioning process between the aqueous phase and the porous medium. Constituents with low  $K_{oc}$  values (i.e., log  $K_{oc}$  less than 3; [Ney, 1990]) are less likely to adsorb to soil. Constituents with log  $K_{oc}$  greater than 4 are likely to adsorb strongly to soil particles.

The  $K_{ow}$  is an indicator of a compound's tendency to partition itself between an organic phase and an aqueous phase. High values of  $K_{ow}$  (e.g., log  $K_{ow}$  greater than 3 [Ney, 1990]) indicate lipophilic compounds which typically bioaccumulate in aquatic organisms and have a greater tendency for adsorption in soils and sediments. Comparative values for  $K_{ow}$  and  $K_{oc}$ , along with log  $K_{oc}$  and  $K_{ow}$  values, were used to rank mobility of the PAH<sub>s</sub>, and are presented in Table 3-2.

Based on rankings of comparative mobility, as measured by  $K_{oc}$  and  $K_{ow}$ , chrysene is considered the most highly mobile of the PAH<sub>s</sub>, followed by benzo(a)pyrene.

#### Relative Toxicity of PAH<sub>s</sub>

The relative carcinogenic toxicity of the PAH<sub>s</sub> was measured by comparison of their cancer slope factors (CSFs). With the exception of B(a)P, CSFs are not available for PAH<sub>s</sub>; however, as an interim procedure, EPA Region IV has adopted a toxicity equivalency factor (TEF) methodology for PAH<sub>s</sub> based on the potency of each compound relative to the potency of B(a)P (USEPA, 1992b). This methodology was not available at the time the RI/FS was completed. TEFs for the seven PAH<sub>s</sub> are shown below:

TABLE 3-2  
CHEMICAL PROPERTIES AND MOBILITY AND TOXICITY RANKING OF COCs

Constituent	MOBILITY INDICATORS (a)				TOXICITY INDICATORS	Kow-Based		Koc-Based		Sum of Kow & Koc Rankings	Overall Mobility Rank (e)	Toxicity Rank (f)	Sum of Mobility & Toxicity Rankings	Overall Mobility & Toxicity Ranking (g)
	log		log			Mobility Rank (d)	Mobility Rank (d)							
	Kow	Koc (b)	Kow	Koc (b)										
Benzo(a)anthracene	7.94E+05	5.90	1.38E+06	6.14	0.1	2	4	6	3	3	6	2		
Benzo(a)pyrene	1.10E+06	6.04	3.98E+05	5.60	1	3	2	5	2	1	3	1		
Benzo(b)fluoranthene	3.72E+06	6.57	5.50E+05	5.74	0.1	5	3	8	4	4	8	5		
Benzo(k)fluoranthene	7.08E+06	6.85	4.37E+06	6.64	0.1	6	6	12	6	5	11	6		
Chrysene	4.07E+05	5.61	2.46E+05	5.39	0.01	1	1	2	1	7	8	4		
Dibenz(a,h)anthracene	2.29E+06	6.36	1.66E+06	6.22	1	4	5	9	5	2	7	3		
Indeno(1,2,3-cd)pyrene	5.01E+07	7.70	3.09E+07	7.49	0.1	7	7	14	7	6	13	7		
Pentachlorophenol	7.24E+05	5.86	9.00E+02	2.95	NA	NA	NA	NA	NA	NA	NA	NA		

(a) Source: PAH values = Montgomery & Welton, 1990; PCP values = USEPA, 1994.

(b) log Kow and log Koc are calculated from Kow and Koc, respectively

(c) Source: USEPA, 1992b

(d) Ranked from highest to lowest mobility, based on Kow or Koc values (lowest Kow or Koc = highest mobility)

(e) Ranked in order of highest to lowest overall mobility, based on sum of Kow and Koc ranking; lowest ranking = highest mobility

(f) Ranked in order of highest to lowest TEF; highest TEF = greatest toxicity

(g) Ranked in order of highest to lowest combined mobility and toxicity, based on sum of mobility and toxicity ranking; lowest ranking = highest combined mobility & toxicity  
NA = Not applicable; pentachlorophenol is not a PAH; it has been selected for evaluation in the VE-RA and does not require ranking for selection.

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Benzo(a)pyrene	1.0
Dibenzo(a,h)anthracene	1.0
Benzo(a)anthracene	0.1
Benzo(b)fluoranthene	0.1
Benzo(k)fluoranthene	0.1
Indeno(1,2,3-cd)pyrene	0.1
Chrysene	0.01

Based on a comparison of the TEF values, B(a)P and dibenzo(a,h)anthracene would be selected as the most toxic of the PAH<sub>c</sub>s.

#### Selection of Indicator PAH<sub>c</sub> for Evaluation

Each PAH<sub>c</sub> was assigned separate scores for mobility and toxicity, according to rank. The scores for each constituent were summed and the constituents were then ranked according to the total score, as shown on Table 3-2. B(a)P received the highest ranking for combined toxicity and mobility and was thus selected as the indicator COC for evaluation in the groundwater modeling and risk analysis.

#### Selection of PCP for Evaluation

In addition to B(a)P, PCP was also selected for evaluation in the VE Risk Analysis. PCP is not a PAH<sub>c</sub>. Its structure, activity, toxic properties, and mechanisms of toxicity differ from those of the PAH<sub>c</sub>s and thus should be evaluated separately. Additionally, although PCP is less toxic than most PAH<sub>c</sub>s, it is more mobile in environmental media. K<sub>ow</sub> and K<sub>oc</sub> values of PCP (shown in Table 3-2) indicate that it is much less likely to be adsorbed to the soil than PAH<sub>c</sub>s.

#### **3.3.2.3.2 Risk Analysis**

In order to compare the effectiveness of the designated remedial action and the no thermal treatment option with respect to overall protection of human health, a comparative risk analysis of the designated and alternative remedial actions was conducted. The comparative analysis incorporates portions of the nine evaluation criteria set forth in the NCP (40 CFR 300.430(e)(9)) which are used to evaluate and

select remedial alternatives, i.e., overall protection of human health and environment (a threshold criterion) and reduction of toxicity, mobility, or volume through treatment (a primary balancing criterion). The potential carcinogenic health risks from residential exposure to COCs in groundwater outside the slurry wall were calculated for each remediation scenario, based on the modeled groundwater concentrations for PCP and B(a)P.

The comparative risk analysis was performed in accordance with the most current EPA guidance for conducting public health risk assessments (USEPA 1991a, 1991b, 1992a). Although the Baseline Risk Assessment (BRA) developed for the site as part of the Remedial Investigation (Dames & Moore, 1992) was performed in accordance with a draft version of the EPA risk assessment guidance, the process remains the same. The risk analysis for each remedial scenario consists of four steps, as follows:

- identification of exposure pathways and receptors for evaluation;
- exposure quantification;
- chemical toxicity assessment; and
- risk estimation and characterization.

With the exception of the exposure point concentrations and resulting risks, the risk analysis is identical for both remediation scenarios evaluated. In addition to these four steps, a discussion of comparison of relative risks associated with each scenario will be presented, along with a discussion of uncertainties.

#### **3.3.2.2.1 Exposure Scenario and Receptors Evaluated**

The goal of the exposure assessment is to estimate the type and magnitude of human exposure to the contaminants present in and migrating from groundwater at the site. The exposure assessment was conducted to estimate the potential risks to off-site receptors from exposure to groundwater under each remedial action scenario. For purposes of relative risk comparison, the most conservative exposure scenarios, i.e., those presenting the greatest exposure durations and chronic intake, were used. The exposure pathways and receptors selected for evaluation are described below.

Under this risk analysis, it was assumed that a small quantity of surface water could infiltrate the soil cap, and that contaminated groundwater may potentially migrate from the site through the slurry wall. As previously described, rates of movement were assumed to be equivalent to the most conservative (i.e., greatest) permeability rates for movement of water through the cap or the slurry wall. Under the most conservative assumption, it was assumed that off-site exposure to groundwater may occur at the location which was shown by the groundwater model to have the greatest constituent concentrations immediately outside the slurry wall in the direction of site groundwater migration. This location, at about the midpoint of the north boundary of the site, was consistent for each aquifer, under each scenario, and was selected as the exposure point. The exposure point is shown on Figure 3-12.

The most conservative pathway of exposure to groundwater at the exposure point, i.e., the pathway that would present the highest levels of exposure, is through residential use of groundwater as potable water. Under such a future residential scenario, residents may potentially install a well at the exposure point in either of the two layers of groundwater. This groundwater could be used in the household for drinking water, and for showering and household cleaning. Thus, exposure to residents via ingestion of drinking water and dermal contact during showering or household use were evaluated in the risk analysis. Volatile organic compounds are not present among the COCs selected for risk analysis. Childhood exposure durations are many years shorter than those of adults. Thus, inhalation of volatilized constituents while showering was considered an incomplete exposure pathway and was not evaluated in this risk analysis.

Adult residents were considered to have the greatest potential exposures to groundwater, as they may be exposed over the duration of their lifetime as a consequence of living at one residence (i.e., in a house receiving water from a well installed at the exposure point). Excess cancer risks are based on lifetime exposure and increase with increasing exposure duration. Thus, adults, rather than children, were selected as the receptors for evaluation.

The pathways and receptors selected for this risk analysis are consistent with those that were evaluated for groundwater exposure in the BRA (Dames & Moore, 1992).



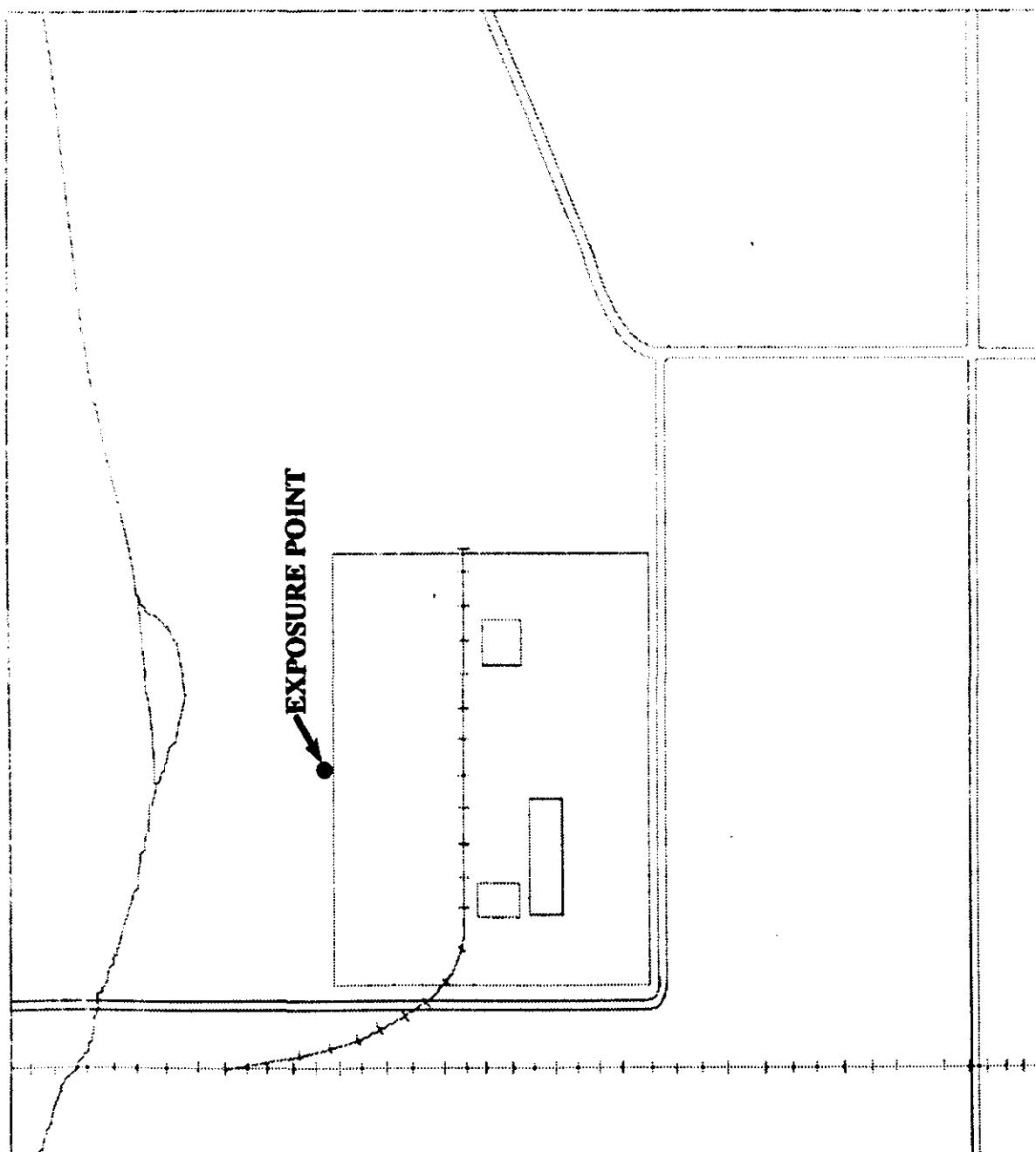


FIGURE 3.12  
LOCATION OF GROUNDWATER EXPOSURE POINT  
FOR RISK ANALYSIS  
VIRGINIA WOOD PRESERVING SITE

#### **3.3.2.2.2.2 Exposure Quantification**

As part of this assessment, the exposure point concentrations, or estimates of the chemical concentrations that will be contacted over time, were selected based on concentrations developed as part of the groundwater flow model described in Section 3.2. For each scenario, the maximum concentration of PCP and B(a)P outside the slurry wall in each aquifer was identified. Concentrations were developed for four different periods of time from remediation: one day, one year, ten years and thirty years. All modeled concentrations for B(a)P were zero for each scenario and aquifer. For each aquifer and each scenario, the modeled maximum concentrations of PCP are greatest after one day, and slowly decrease over time. In the perched aquifer, modeled concentrations of PCP under both remediation scenarios at one day and one year were 2.0 and 1.9  $\mu\text{g/l}$ , respectively, and at ten and thirty year time frames, the aquifer was dry. The modeled concentrations of PCP in the saprolite aquifer were 15.0, 14.9, 14.2 and 10.4  $\mu\text{g/l}$  after one day and one, ten and thirty years, respectively, under both remediation scenarios. These results indicate that, in each given time frame, there is no difference in maximum groundwater concentrations of PCP at the exposure point under either remediation scenario.

The modeled concentrations at the north boundary at the one year time frame were selected for use in the risk analysis. Although there is very little difference in the groundwater concentrations over time, the use of concentrations modeled at this time frame is more conservative than using concentrations at ten and thirty years, yet is more realistic than using concentrations at one day after remediation. The selected exposure point concentrations under each remediation scenario for each COC are presented in Table 3-3, below, for each aquifer layer. Although modelling has shown slight decreases in groundwater concentrations, for purposes of this risk analysis they are assumed to remain constant throughout the entire exposure period, i.e., no degradation or attenuation of constituents is assumed to occur.

**TABLE 3-3**  
**Exposure Point Concentrations**  
**Used in Risk Analysis**

	Constituent Concentrations (1 year)	
	Pentachlorophenol (mg/l)	Benzo(a)pyrene (mg/l)
<b>Scenario 1 (LTDD)</b>		
Perched Aquifer	0.0019	0.0
Saprolite Aquifer	0.0149	0.0
<b>Scenario 2 (no LTDD)</b>		
Perched Aquifer	0.0019	0.0
Upper Aquifer	0.0149	0.0

Pathway-specific intakes were calculated for each COC using EPA recommended methods (USEPA, 1991a). An intake incorporates the exposure point concentration and rate of contact with the contaminated groundwater, and expresses exposure in terms of mass of a contaminant to which an individual is exposed per body weight per unit time (e.g., mg/kg/day). The intake is an estimate of chemical-specific exposure that will be used to calculate risk for each contaminant within an exposure pathway. The parameters used to characterize contact rate and subsequent exposure intake via ingestion of drinking water and dermal exposure while showering are based on standard EPA default values for a reasonable maximum exposure. These exposure parameters and resulting exposure intakes for drinking water ingestion and dermal contact with water are presented in Tables 3-4 and 3-5, respectively, and are described below.

The pathway-specific intake calculations incorporate upper-bound and mid-range exposure factors, representing an exposure scenario that is both protective and reasonable; not the worst possible case. Both pathways incorporate an exposure frequency of 350 days/year, which assumes the resident spends two weeks away from home per year (USEPA, 1991b). An exposure duration of 30 years, which is both the 90th percentile of time spent at one residence (USEPA, 1991b) and the estimated

**TABLE 3-4**  
**FUTURE RESIDENTIAL EXPOSURE**  
**INGESTION OF GROUNDWATER USED AS POTABLE WATER**  
**INGESTION INTAKES**

$\text{Ingestion Intake} = \frac{C \times CR \times ET \times EF \times ED}{BW \times AT}$ <p>(mg/kg-day)</p> <p>where:</p> <p>C = Concentration of Constituents in Groundwater (mg/L)</p> <p>IR = Ingestion Rate (L/day)</p> <p>EF = Exposure Frequency (days/year)</p> <p>ED = Exposure Duration (years)</p> <p>BW = Body Weight (kg)</p> <p>AT = Averaging Time (days)</p>	
Exposure Variable	Adult
CR	2.0
EF	350
ED	30
BW	70
AT (Carcinogens)	25,550
<b>PATHWAY-SPECIFIC INTAKE<sup>1</sup> = C (mg/L) x 1.17x10<sup>-2</sup> L/kg-day</b>	

<sup>1</sup>Chemical-specific intakes for each remediation scenario and each aquifer are presented in the risk calculation sheet for groundwater ingestion in Tables 3-E.

TABLE 3-5

**FUTURE RESIDENTIAL EXPOSURE  
DERMAL EXPOSURE TO GROUNDWATER  
DERMAL UPTAKES**

Dermal Uptake = $\frac{C \times SA \times PC \times ET \times EF \times ED \times CF}{BW \times AT}$ (mg/kg-day)	
where: <ul style="list-style-type: none"> <li>C = Concentration of Constituents in Groundwater (mg/L)</li> <li>SA = Surface Area of Exposed Skin (cm<sup>2</sup>)</li> <li>PC = Permeability Constant (cm/hour)</li> <li>ET = Exposure Time (hours/day)</li> <li>EF = Exposure Frequency (days/year)</li> <li>ED = Exposure Duration (years)</li> <li>CF = Conversion Factor (L/cm<sup>3</sup>)</li> <li>BW = Body Weight (kg)</li> <li>AT = Averaging Time (days)</li> </ul>	
Exposure Variable	Adults
SA	21,500
PC	Chemical-Specific
ET	0.25
EF	350
ED	30
CF	0.001
BW	70
AT (Carcinogens)	25,550
PATHWAY-SPECIFIC UPTAKE <sup>2</sup> = $C(\text{mg/L}) \times PC(\text{cm/hour}) \times 3.16 \times 10^{-2} \text{ L-hr/(cm-kg-day)}$	

<sup>2</sup>Chemical-specific intakes for each remediation scenario and each aquifer are presented in the risk calculation sheets for dermal contact in Table 3-F.

lifetime of the slurry wall, is assumed. An EPA standard default adult body weight of 70 kg is used (USEPA, 1991b). The averaging time term represents the time period over which the exposure is estimated. For carcinogens, intakes are calculated by prorating the total cumulative dose over an entire lifetime. The EPA default value for an average lifetime is 70 years (USEPA, 1991b).

Pathway-specific assumptions are described as follows. For groundwater ingestion, a daily ingestion rate of 2 liters/day, which is the 90th percentile (upper-bound) value for drinking water ingestion (USEPA, 1991b), was used in the intake calculations. For dermal contact, the intake calculation incorporates the surface area (SA) of exposed skin and an exposure time (ET). The SA of 21,500 cm<sup>2</sup> represents the upper-bound total adult body surface area which is exposed to water while showering (USEPA, 1992a), and the ET of 0.25 hours represents an upper-bound estimate of time spent in the shower each day (USEPA, 1992a). Chemical-specific dermal permeability constants used in the equation for dermal uptake ( $K_p$  or PCs) are obtained from the EPA's dermal exposure assessment guidance, Dermal Exposure Assessment: Principles and Applications (USEPA, 1992a).

### **3.3.2.2.2.3 Toxicity Assessment**

The purpose of the toxicity assessment is to compile and evaluate noncarcinogenic and carcinogenic toxicity data for the chemicals of concern. The assessment provides an estimate of the relationship between the extent of potential contaminant exposure and the increased likelihood and/or severity of adverse effects (i.e., the dose-response relationship). As previously described, the reduction of cancer risks associated with PCP and PAHs was the primary consideration in the selection of LTDD as a remedy. Since noncarcinogenic health effects were not considered in the selection process, they were not evaluated in the VE Risk Analysis. Thus, only carcinogenic toxicity data were included in the Toxicity Assessment.

Summaries of the qualitative human health toxicity information for PCP and B(a)P are presented in Appendix C-B of the BRA conducted for the site (Dames & Moore, 1992) and will not be repeated here. The quantitative human health toxicity criteria which

were used in the BRA have been updated. Dames & Moore gathered toxicity criteria for PCP and B(a)P available as of October 1994 (USEPA, 1994). The cancer slope factors (CSFs) for acceptable intakes for chronic exposures are quantitative values of toxicity used to estimate potential human carcinogenic risks. CSFs for oral exposure are used to estimate risks via both ingestion and dermal contact. The available oral CSFs are  $1.21 \times 10^{-1} \text{ (mg/kg-day)}^{-1}$  for PCP and  $7.3 \text{ (mg/kg-day)}^{-1}$  for B(a)P. These values were used to calculate excess cancer risks associated with exposure to these constituents (USEPA, 1994).

#### 3.3.2.2.4 Risk Characterization

The risk characterization was conducted using standard EPA methods, which were described in detail in the BRA conducted for the site (Dames & Moore, 1992). The calculations of excess cancer risks and results for each exposure pathway are presented in Tables 3-6 and 3-7. Since the modeled B(a)P concentrations for both remediation scenarios were zero in each aquifer at all points outside the slurry wall, there was no excess cancer risk associated with this constituent. The calculated risks are attributed entirely to exposure to PCP. For the ingestion pathways, the excess cancer risks calculated for PCP in the perched aquifer were  $2.7 \times 10^{-6}$  in both Scenarios 1 and 2. The risks from ingestion of PCP in groundwater from the saprolite layer were  $2.1 \times 10^{-6}$  in both Scenarios 1 and 2. For dermal exposure, PCP in the perched aquifer was associated with an excess cancer risk of  $4.7 \times 10^{-6}$  in both remediation scenarios. Likewise, dermal contact with PCP in saprolite groundwater was associated with a risk of  $3.7 \times 10^{-6}$  for both scenarios.

The risks from each pathway are combined to give a total excess cancer risk from groundwater exposure for the adult receptor. A summary of the total risks for each scenario, by aquifer, are presented in Table 3-8.

**TABLE 3-6**  
**EXCESS CANCER RISKS**  
**RESIDENTIAL INGESTION OF GROUNDWATER**

Parameter	Exposure Point Conc. (mg/L)	Carcinogen Intake Factor (L/kg-day)	Chronic Intake (mg/kg-da)	Cancer Slope Factor (kg-day/m)	Excess Cancer Risk (unitless)
<b>SCENARIO 1 (LTDD)</b>					
<b>UPPER LAYER AQUIFE</b>					
Benzo(a)pyrene	0	1.17E-02	0.00E+00	7.30E+00	0.00E+00
Pentachlorophenol	0.0019	1.17E-02	2.23E-05	1.21E-01	2.70E-06
<b>TOTALS:</b>					<b>2.7E-06</b>
<b>SAPROLITE AQUIFER</b>					
Benzo(a)pyrene	0	1.17E-02	0.00E+00	7.30E+00	0.00E+00
Pentachlorophenol	0.0149	1.17E-02	1.75E-04	1.21E-01	2.12E-05
<b>TOTALS:</b>					<b>2.1E-05</b>
<b>SCENARIO 2 (SOILS LEFT IN PLACE)</b>					
<b>UPPER LAYER AQUIFE</b>					
Benzo(a)pyrene	0	1.17E-02	0.00E+00	7.30E+00	0.00E+00
Pentachlorophenol	0.0019	1.17E-02	2.23E-05	1.21E-01	2.70E-06
<b>TOTALS:</b>					<b>2.7E-06</b>
<b>SAPROLITE AQUIFER</b>					
Benzo(a)pyrene	0	1.17E-02	0.00E+00	7.30E+00	0.00E+00
Pentachlorophenol	0.0149	1.17E-02	1.75E-04	1.21E-01	2.12E-05
<b>TOTALS:</b>					<b>2.1E-05</b>

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**TABLE 3-7**  
**EXCESS CANCER RISKS**  
**RESIDENTIAL DERMAL EXPOSURE TO GROUNDWATER WHILE SHOWERING**

Parameter	Exposure Point Conc. (mg/L)	Chemical-Specific PC (cm/hr)	Carcinogen Intake Factor (L/kg-day)	Chronic Intake (mg/kg-da)	Cancer Slope Factor (kg-day/m)	Excess Cancer Risk (unitless)
<b>SCENARIO 1 (LTDD)</b>						
<b>PERCHED AQUIFER</b>						
Benzo(a)pyrene	0	1.2	3.16E-02	0.00E+00	7.30E+00	0.00E+00
Pentachlorophenol	0.0019	0.65	3.16E-02	3.90E-05	1.21E-01	4.72E-06
<b>TOTALS:</b>						<b>4.7E-06</b>
<b>SAPROLITE AQUIFER</b>						
Benzo(a)pyrene	0	1.2	3.16E-02	0.00E+00	7.30E+00	0.00E+00
Pentachlorophenol	0.0149	0.65	3.16E-02	3.06E-04	1.21E-01	3.70E-05
<b>TOTALS:</b>						<b>3.7E-05</b>
<b>SCENARIO 2 (SOILS LEFT IN PLACE)</b>						
<b>UPPER LAYER AQUIFER</b>						
Benzo(a)pyrene	0	1.2	3.16E-02	0.00E+00	7.30E+00	0.00E+00
Pentachlorophenol	0.0019	0.65	3.16E-02	3.90E-05	1.21E-01	4.72E-06
<b>TOTALS:</b>						<b>4.7E-06</b>
<b>SAPROLITE AQUIFER</b>						
Benzo(a)pyrene	0	1.2	3.16E-02	0.00E+00	7.30E+00	0.00E+00
Pentachlorophenol	0.0149	0.65	3.16E-02	3.06E-04	1.21E-01	3.70E-05
<b>TOTALS:</b>						<b>3.7E-05</b>

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**TABLE 3-8  
SUMMARY OF EXCESS CANCER RISK BY SCENARIO AND AQUIFER**

Exposure Pathway	Excess Cancer Risk (unitless)			
	Perched Aquifer		Saprolite Aquifer	
	Scenario 1	Scenario 2	Scenario 1	Scenario 2
Groundwater ingestion	$2.7 \times 10^{-6}$	$2.7 \times 10^{-6}$	$2.1 \times 10^{-5}$	$2.1 \times 10^{-5}$
Dermal Contact	$4.7 \times 10^{-6}$	$4.7 \times 10^{-6}$	$3.7 \times 10^{-5}$	$3.7 \times 10^{-5}$
Total Receptor Risk:	$7.4 \times 10^{-6}$	$7.4 \times 10^{-6}$	$5.8 \times 10^{-5}$	$5.8 \times 10^{-5}$

As shown in Table 3-G, the total excess cancer risk from exposure to PCP in perched groundwater under Scenario 1 is  $7.4 \times 10^{-6}$ . This is equivalent to the total excess cancer risk from exposure to perched groundwater under Scenario 2. The total excess cancer risk from exposure to saprolite groundwater is  $5.8 \times 10^{-5}$  under both Scenarios 1 and 2. Therefore, using PCP and B(a)P as surrogate chemicals, the inclusion of LTDD in the remedial action has no effect on the potential cancer risk posed by exposure to groundwater leaving the containment area.

#### **3.3.2.2.2.5 Perspective**

The human health risks from exposure to PCP in perched and saprolite groundwater provide a reference point for comparing the benefits of remedial alternatives. They do not represent actual risks, as the groundwater migrating from the site is not used as potable water. Although the perched and saprolite groundwater units both have the characteristics of a Class II-A aquifer, future domestic use of the groundwater from these units is not likely to occur, since a public water source is available.

#### **3.3.2.2.2.6 Summary of Comparative Analysis of Alternatives**

The designated remedial action of removal of the soil from contaminated areas and treatment with LTDD (i.e., Scenario 1) was selected in the ROD (Dames & Moore,

1992) because it satisfied the nine evaluation criteria set forth in the NCP at 40 CFR 300.430(e)(9). These nine criteria are categorized below in three groups: threshold criteria, primary balancing criteria, and modifying criteria. Each criterion is defined in the ROD for the site (Dames & Moore, 1992).

**THRESHOLD CRITERIA**

1. Overall protection of human health and the environment
2. Compliance with applicable or relevant and appropriate requirements (ARARs)

**PRIMARY BALANCING CRITERIA**

3. Long-term effectiveness and permanence
4. Reduction of toxicity, mobility, or volume through treatment
5. Short-term effectiveness
6. Implementability
7. Cost

**MODIFYING CRITERIA**

8. State acceptance
9. Community acceptance

A comparative analysis of excess cancer risks associated with an alternative remediation strategy (Scenario 2) demonstrates that, with respect to evaluation criteria 1-5, the inclusion of LTTD in the remedial action results in no additional benefit. However, the designated use of LTTD will increase the cost of the remedial action considerably. In the following discussion, the criteria pertaining to human health protection and toxicity reduction (i.e., Criteria 1 through 5) and implementability and cost (Criteria 6 and 7) are presented, along with a comparison of the benefits of the remediation scenarios with respect to each criterion.

**Criterion 1. Overall Protection of Human Health and the Environment**

A primary requirement of CERCLA is that the selected remedial action be protective of human health and the environment. A remedy is considered protective if it reduces current and potential risks to acceptable levels within the established risk range posed by each exposure pathway at the site. The overall risks from groundwater exposure under each scenario are equivalent. Thus, compared with leaving the soil in place without treatment, no further reductions in groundwater concentrations and associated cancer risks from groundwater exposure will be achieved with LTTD treatment of the soil. Therefore, Scenario 2 is equally protective of human health.

## **Criterion 2. Compliance with ARARs**

This criterion addresses whether a remedy will meet all of the ARARs or other environmental statutes and/or provide grounds for invoking a waiver under the NCP (40CFR 300.430(f)(1)(ii)(C)). The ROD for the site states that alternatives must meet all of the respective ARARs of federal and Virginia law. Since the exposure point concentrations in groundwater migrating through the slurry wall are identical for both the designated remedial action and the proposed alternative, no additional benefit in terms of compliance with ARARs for groundwater would be achieved from LTDD treatment of the soil. However, it has become clear from analysis that proceeding with LTDD with a non-combustive APC system poses serious questions regarding compliance with ARARs as a result of the generation of fugitive dust during excavation and potential problems meeting RCRA Subpart O requirements for air emissions.

## **Criterion 3. Long-Term Effectiveness and Permanence**

This evaluation criterion addresses the long-term protection of human health and the environment once remedial action cleanup goals have been achieved, and focuses on residual risks that will remain after completion of the remedial action. The risk characterization was conducted using modeled groundwater exposure concentrations during the first year following remediation. In both aquifers, the groundwater concentrations for Scenario 1 are equivalent to those modeled under Scenario 2. As previously described, this is also true for the modeled concentrations 10 and 30 years after remediation implementation. Equivalent concentrations will also result in equivalent cancer risks associated with groundwater exposure. Thus, the long-term effectiveness of remediation in terms of reduction of cancer risks from groundwater exposure is equivalent under both Scenarios 1 and 2, and there is no increased effectiveness achieved from LTDD treatment of soil. This conclusion is consistent with the ROD for the site, which states that each of the remediation alternatives provides long-term effectiveness and permanence.

## **Criterion 4. Reduction of Toxicity, Mobility, or Volume through Treatment**

This evaluation criterion addresses the degree to which a technology or remedial alternative reduces the toxicity, mobility, or volume of a hazardous substance. The Feasibility Study conducted during the site RI/FS and the development of the ROD (Dames & Moore, 1992; USEPA, 1993) have demonstrated that reduction of the

toxicity and volume of organic constituents in groundwater on-site (and hence, off-site) will be achieved through the implementation of a passive groundwater collection and treatment system that is common to both remediation scenarios. The results of the groundwater modeling and risk analysis for each remediation scenario further demonstrate reductions in the toxicity, mobility and volume of chemicals in groundwater leaving the containment area. However, the identical results of these analyses demonstrate that the use of LTTD in the remedial action will not result in any additional reduction of toxicity, mobility and volume of constituents in groundwater moving from the site. Also, it is clear that LTTD is not actually a destructive treatment technology, yet is merely a volume reducing technology which relies on off-site incineration for ultimate treatment.

#### **Criterion 5. Short-Term Effectiveness**

This evaluation criterion addresses the period of time needed to achieve protection of human health and the environment, and any adverse impacts that may be posed during the construction and implementation period of a remedy, until cleanup goals are achieved. Installation of the cap, slurry wall and groundwater recovery system (Scenario 2) would take approximately twelve to eighteen months to implement. Implementation of the LTTD action (Scenario 1) would take approximately twelve to eighteen months in addition to the time needed for Scenario 2. The estimated time for completion of each remedial action does not include the time for long-term groundwater monitoring, which will be required for each alternative. The amount of soil excavation required and thus, the amounts of dust production, noise disturbance, and truck traffic, would also be significantly increased under Scenario 1. The identical results of the comparative risk analysis for each scenario demonstrate that there is no added benefit associated with the increased soil disturbance and additional four months of implementation time connected with the LTTD remedial action.

#### **Criterion 6. Implementability**

This evaluation criterion addresses the technical and administrative feasibility of each remedy, including the availability of materials and services needed to implement the chosen remedy. The ROD demonstrates that both remedial scenarios meet this criterion, although the LTTD action requires a considerable amount of effort beyond that required for Scenario 2. Scenario 2 is thus the more implementable alternative.

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The identical results of the risk analysis for each scenario, however, do not demonstrate any added benefit in terms of the preceding criteria to warrant the implementation of LTDD treatment at the site.

#### **Criterion 7. Cost**

Section 121 of CERCLA, 42USC Section 9621, requires selection of a cost-effective remedy that protects human health and the environment and meets the other requirements of the statute. A remedy is considered cost-effective if it mitigates the risks posed by the site contamination within a reasonable period of time. Cost-effectiveness is evaluated by first determining whether the remedy satisfies the threshold criteria of protection of human health and the environment and compliance with ARARs (Criteria 1 and 2). As previously described, both remediation scenarios meet these criteria equally. The effectiveness of the alternative is then determined by evaluating the following three of the five balancing criteria: long-term effectiveness and permanence (Criterion 3), reduction of toxicity, mobility, or volume through treatment (Criterion 4), and short-term effectiveness (Criterion 5). As previously demonstrated, each scenario is associated with equal health risks and is equally effective under Criteria 3 and 4. Scenario 2 (no LTDD action) will take less time to implement and involve less soil disturbance, and is thus more effective under Criterion 5.

The remedial scenarios are compared with respect to present worth cost, which includes all capital costs and the operation and maintenance cost incurred over the life of the project. Capital costs include those expenditures necessary to implement a remedial action, including construction costs. Scenario 2 has the lowest present worth cost, estimated at approximately \$9,000,000. The estimated present worth cost of Scenario 1 is approximately \$16,000,000. Based on the preceding evaluation criteria and comparisons of risk and benefits, no improvement in the excess cancer risk associated with off-site groundwater exposure will be achieved through the implementation of LTDD, and thus, there is no substantial basis to support the added expense of \$7,000,000 to implement this remedy.

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#### **3.3.2.2.7 Conclusions**

The results of the modeling and risk analysis for groundwater migrating off-site under the designated remedial action (Scenario 1) and the proposed remedial alternative (Scenario 2) are identical for each aquifer. These results were presented for evaluation under seven of nine NCP criteria for selection of remedial alternatives, which were used to select Scenario 1 as the designated action in the ROD. The excess cancer risks associated with exposure to PCP in groundwater under each scenario, even when using the most conservative criteria, are identical; thus, both scenarios meet the two threshold criteria (protection of human health and the environment and compliance with ARARs) and the first two primary balancing criteria (long-term effectiveness and permanence, and reduction of toxicity, mobility, or volume through treatment) equally. Evaluation under the last three primary balancing criteria (short-term effectiveness, implementability, and cost) demonstrates that Scenario 2 is preferred. This remedial action scenario would take less time to implement, would be associated with less soil disturbance and site impact (e.g., noise disturbance, truck traffic and dust migration), is more implementable, and costs significantly less than the designated remedial action. Based on the analysis of the surrogate chemicals, the implementation of LTTD will not result in a reduction of excess cancer risk from exposure to groundwater migrating off-site. Thus, there is no demonstrated advantage to the increased amount of time, money, and effort that would be required to implement the LTTD remedial action.

#### **3.3.2.3 Air Pollution Control System Analysis**

The ROD specifies that the LTTD system must meet ARARs associated with the site regarding air pollution control. The most important of these regulations are RCRA Subpart O, which essentially requires thermal treatment systems to achieve a 99.99% Destruction Removal Efficiency (DRE) for Principal Organic Hazardous Constituents (POHCs) and the Best Available Control Technology (BACT) regulations promulgated by the Commonwealth of Virginia.

Based on treatability analysis from the RD Work Plan, it is clear that achieving a 99.99% DRE in a thermal desorber for the constituents of concern is questionable at best. Trying to meet this requirement, or any other discharge requirement will necessitate the need for an air pollution control system (non-combustive) that will by its nature generate numerous waste streams including:

- pure product organics;
- contaminated wastewater;
- carbon from water treatment;
- carbon from air stream treatment; and
- air stream.

Complicating issues and greatly increasing the cost of this process is the ultimate disposal of these waste streams. All of these waste streams except the air stream will contain regulatory-significant concentrations of dioxin, which result in extremely high disposal costs (assuming that these materials can be disposed of at all).

An alternative to the non-combustive air pollution system is the use of an oxidative combustion unit for air pollution control. Using this system would allow for the elimination of all waste streams, thus greatly reducing the cost of LTTD. The Commonwealth of Virginia has expressed that oxidative combustion is the BACT for thermal treatment systems. This would indicate that this is the preferred technology by the state for this solution.

It is recommended that based on regulatory compliance concerns and cost considerations that oxidative combustion be given serious consideration as the APC of choice for this RA.



## **4.0 VALUE ENGINEERING CONCLUSIONS/RECOMMENDATIONS**

### **4.1 GROUNDWATER EXTRACTION**

Consideration should be given to the use of a interceptor trench system for perched aquifer groundwater extraction while utilizing the caissons for saprolitic groundwater extraction.

### **4.2 WATER TREATMENT**

The regulatory situations regarding water discharge standards and the effect of dioxin in this discharge should be evaluated. UV/Oxidation should be considered as a replacement for carbon adsorption for organics treatment. Cost/Benefit analysis should be performed and regulatory issues resolved regarding on-site disposal of treatment of spent carbon and process sludge.

### **4.3 SOIL TREATMENT**

A serious reevaluation of the necessity of LTDD on this site based on regulatory, risk and cost elements should be considered. This is most certainly a good candidate for full VE analysis. If LTDD remains the remedy for the site, a study of combustive versus non-combustive APC systems regarding cost and technical applicability should be conducted.

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